UNITED NATIONS

Food and Agriculture Organization of the United Nations

BES

IPBES/6/INF/4/Rev.1*

Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services

Distr.: General
22 June 2018

English only

Plenary of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services
Sixth session
Medellin, Colombia, 18–24 March 2018
Agenda item 6 (b)

Regional and subregional assessments of biodiversity and ecosystem services: regional and subregional assessment for the Americas

Chapters of the regional and subregional assessment of biodiversity and ecosystem services for the Americas

Note by the secretariat

1. In paragraph 2 of section III of decision IPBES-3/1, the Plenary of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) approved the undertaking of four regional and subregional assessments of biodiversity and ecosystem services for Africa, the Americas, Asia and the Pacific, and Europe and Central Asia (hereinafter referred to as regional assessments) in accordance with the procedures for the preparation of the Platform's deliverables set out in annex I to decision IPBES-3/3, the generic scoping report for the regional assessments of biodiversity and ecosystem services set out in annex III to decision IPBES-3/1, and the scoping reports for each of the four regional assessments (decision IPBES-3/1, annexes IV–VII).

2. In response to decision IPBES-3/1, a set of six chapters (IPBES/6/INF/3–6), together with a summary for policymakers (IPBES/6/4–7), were produced for each of the regional assessments by an expert group, in accordance with the procedures for the preparation of the Platform's deliverables, for consideration by the Plenary at its sixth session.

3. In paragraph 5 of section IV of decision IPBES-6/1, the Plenary approved the summary for policymakers of the regional assessment for the Americas (IPBES/6/15/Add.2) and accepted the chapters of the assessment, on the understanding that the chapters would be revised following the sixth session as document IPBES/6/INF/4/Rev.1 to correct factual errors and to ensure consistency with the summary for policymakers as approved. The annex to the present note, which is presented without formal editing, sets out the final set of chapters of the assessment for the Americas including their executive summaries.

4. A laid-out version of the final regional assessment report of biodiversity and ecosystem services for the Americas (including a foreword, statements from key partners, acknowledgements, a preface, the summary for policymakers, the revised chapters and annexes setting out a glossary and lists of acronyms, authors, review editors and expert reviewers) will be made available on the website of the Platform prior to the seventh session of the Plenary.

* Reissued for technical reasons on 24 September 2018.

K1802537 240918

Annex

Chapters of the regional assessment report on biodiversity and ecosystem services for the Americas of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services

Disclaimer on maps

The designations employed and the presentation of material on the maps used in this report do not imply the expression of any opinion whatsoever on the part of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystems Services concerning the legal status of any country, territory, city or area or of its authorities, or concerning the delimitation of its frontiers or boundaries. These maps have been prepared for the sole purpose of facilitating the assessment of the broad biogeographical areas represented therein.

Table of contents

Chapter 1: Setting the scene

Coordinating Lead Authors:
Jake Rice (Canada), Vanesa Rodríguez Osuna (Bolivia/USA), Maria Elena Zaccagnini (Argentina)

Lead Authors:
Elena Bennett (Canada), Dayne Buddo (Jamaica), Natalia Estrada-Carmona (Colombia/France), Kelly Garbach (USA), Nathan Vogt (Brasil/USA)

Fellow
María Paula Barral (Argentina)

Contributing Authors
Judith Weis (USA), Cristiana Simão Seixas (Brazil), Rosely Sanches (Brazil), Mary Kalin Arroyo (Chile)

Review Editors
Patricia Balvanera (Mexico), Rodolfo Dirzo (Mexico)

To be cited as:
Rice, J., Rodríguez Osuna., V., Zaccagnini, M. E., Bennet, E. Buddo, D., Estrada-Carmona, N., Garbach, K., Vogt, N., and Barral, M. P. Chapter 1: Setting the scene. In IPBES (2018): The IPBES regional assessment report on biodiversity and ecosystem services for the Americas. Rice, J., Seixas, C. S., Zaccagnini, M. E., Bedoya-Gaitán, M., and Valderrama, N. (eds.). Secretariat of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services, Bonn, Germany, pp. xx-xx

Table of contents Chapter 1

1 Executive summary

1. **The Americas region is highly biologically diverse, hosting seven of the 17 most biodiverse countries of the world and encompassing 14 units of analysis across 140 degrees of latitude {1.1}**. The Americas include 55 of the 195 terrestrial and freshwater world ecoregions with highly distinctive or irreplaceable species composition. The region is home to 20 per cent of globally identified key biodiversity areas, 26 per cent of global terrestrial biodiversity hotspots, and the Gulf of California and Western Caribbean are included in the top 18 marine biodiversity conservation hotspots. The region also has some of the most extensive wilderness areas in the planet, such as the Pacific Northwest, the Amazon, and Patagonia, and contains three of the six longest coral reefs in the world.

2. **The Americas is also culturally and socio-economically diverse,** home to some of the most industrialized urban areas on the planet and to indigenous and other local people striving to maintain and protect their cultures. The region is populated by a uniquely large proportion of immigrants (and their descendants) from all parts of Europe, Asia and Africa, in addition to the more than 66 million indigenous peoples who have persisted despite centuries of land expropriation and, in some cases, active persecution and even genocide. Human population density in the Americas ranges from 2 per100 km^2 in Greenland to over 9,000 per km^2 in several urban centers. The Americas region contains two of the ten countries with the highest Human Development Index in the world as well as one with the lowest human development level {1.6.1-1.6.3}.

3. **Ecosystems in the Americas provide essential contributions to the economy, livelihoods, food, water, and energy security, and to the eradication of poverty in the region**. **Increases in the use of nature has resulted in the region being the largest global exporter of food.** People's quality of life in the Americas is highly dependent on nature's material contributions (including food and feed, medicine, energy, fibers, and construction materials) to achieve food, water and energy security, and to generate income and support livelihoods and health. The Americas is an important commodity producer: countries of the Americas are amongst the top 10 producers (in terms of volume in 2014 and 2015) of wheat, rice, sugar, coarse grains, tea, coffee, cocoa, and orange juice. Several countries are important producers of aquaculture and fisheries in terms of volume of fish, crustaceans and molluscs harvested in 2014. The United States of America and Brazil are the second and third largest meat producers (in terms of volume in 2013 or 2014). These two countries in addition to Argentina are the world's top three major oil seed (soybeans, rapeseed, cottonseed, sunflower seed and groundnuts) producers (in terms of volume in 2014 to 2015) {1.3.2}. The region has a mosaic of indigenous, small-scale, and large-scale agriculture production, which builds on a foundation of the biodiverse American tropics and montane regions. These regions are major centers of origin for domesticated plants, some of which have subsequently become important globally-traded crops {1.1}.

4. **Forests and wetlands are the ecosystems mostly recognized for their role in the regulation of freshwater supplies, which is abundant (compared to the global average) but unevenly available across geographies and time.** Some cities in South America face severe water scarcity episodes during specific times of the year (Bogotá, Quito, La Paz, Lima) as well as in states of the United States of America such as California, Texas and Florida. Areas with high scarcity occur where densely populated areas compete with intensely irrigated agriculture, or areas with reduced water storage. Climate change impacts and unsustainable rates of extraction of freshwater result in reduced river flows as in the Colorado River. Groundwater depletion also occurs in the Americas (Mexico and United States of America), affecting water users, business operations, and biodiversity {1.2.1,1.3.2}.

5. **Trends in livelihoods and good quality of life depend not only on material nature contributions to people (e.g. fish, food, fiber) with high economic value, but also on non-material nature's contributions to people (e.g. learning and experiences, supporting identities) and regulating (e.g. regulation of extreme events, disease, pollination) that often are not accounted for in traditional**

economic measures {1.3.2}. The perception of nature's contributions to people depends on a person's worldview. Nature's non-material contributions help societies achieve a compassionate and equitable life by providing opportunities for learning and inspiration for culture, as well as helping form identity, social cohesion, and symbolic bonds with nature.

6. **There is considerable evidence of the harmful effects of nature's degradation on public health, livelihoods and both regional and national economies in the Americas {1.2.1}**. The harmful effects of nature degradation (e.g. air and water pollution, deforestation) disproportionately affect the poorest populations and therefore pose a threat to inclusive development {1.3.1}. The degradation of nature frequently involves the loss of (natural) assets, which are typically not taken into account in traditional economic measures. Thus, a country may deplete a natural resource base (e.g. forests) to provide positive economic gains even as that resource depletion has other, unaccounted-for consequences, such as degrading regulating contributions (e.g. water supply) and non-material contributions to good quality of life, including recreation, spirituality, religion, and identity.

7. **Agricultural production has increased its footprint through the extensification (spreading to new areas), and intensification (greater use of technologies), producing elevated nutrient loading, and introducing pesticides and other agrichemicals into ecosystems.** These elevated levels of nutrients and pollutants have negative consequences for ecosystem function, and air, soil and water quality, including major contributions to coastal and freshwater oxygen depletion creating "dead zones" with impacts on biodiversity, human health, and commercial fisheries {1.2.1}.

8. **The plurality of values in the Americas shapes use, management and conservation of nature and nature's contributions to people {1.1}.** In particular, trade-offs are experienced in different ways by people holding different worldviews and cultures, depending on their values {1.1}. Regional differences can also influence the way policies affect value given to ecosystems {1.2.4}. Policies addressing ecotourism could emphasize the substantial economic benefits from recreational use associated with ecotourism in conserved areas or give more weight to protective approaches to biodiversity conservation and restrict ecotourism stringently {1.5.5}.

9. **All policies can affect nature's health, and thus its contributions to people, by altering positively and negatively how governments, institutions, and individuals interact with people and nature through regulation, incentive mechanisms, and rights-based approaches {1.5.5}.** Benefits from policies providing incentives for increasing or protecting some elements of nature, if not designed and implemented carefully, bring costs of in the loss or reduction of other aspects of nature or nature's contributions to people. For example, the creation of protected areas may come at the cost of displacement of local community uses of the areas, such as when marine protected areas attract significant ecotourism revenues, but displace community-based fisher families with few alternative options for livelihoods. Policies can also provide purposeful or incidental disincentives to using nature and nature's contributions to people responsibly provide disincentives to use nature and nature's contributions to people responsibly. For example, in the energy sector, domestic subsidies of fuel prices promote overutilization of these resources, increase greenhouse gas emissions, which have a negative contribution to climate change accelerating its impacts on biodiversity and people. Alternative policies such as carbon tax or eliminating subsidies for producing or consuming fossil fuels may have different consequences, including improving energy efficiency, development of renewable energy sources and generating health benefits for people. However, such alternatives must be considered fully, as hydroelectric power may require substantial modifications to natural watersheds, and mining the raw materials needed for solar panels can have a large environmental footprint.

10. **These trade-offs highlight the complexities that exist in developing responsible policies for conservation and sustainable use of nature and nature's contributions to people and the importance of the efforts of the Intergovernmental Panel on Biodiversity and Ecosystem Services Regional Assessments to consider the multiple knowledge systems and the values of diverse worldviews,**

including the use of scenarios and models effectively {1.5.5}. The effectiveness and impact of policies and interventions related to nature's components depend on the way societies perceive the world, negotiate interests, prioritize problems, and find feasible solutions that respect social, institutional, and environmental settings. Such enabling conditions are essential to foster a successful implementation of policies that include environmental and other societal issues (e.g. poverty reduction, including local knowledges and minorities).

11. The objectives of this Assessment are to: a) evaluate the contribution of biodiversity and ecosystem functions and services to the economy, livelihoods, food security, and good quality of life in the Americas; b) identify major trends of biodiversity and ecosystems (nature) and ecosystem functions and services, as nature's contributions to people; c) assess the implications of these trends for human well-being (quality of life) experienced by various societies and cultures; d) identify future potential threats to biodiversity and ecosystems (nature) as well as the nature's contributions to people that they provide) and the implications of the threats for a good quality of life; and e) identify opportunities for avoiding or mitigating threats to biodiversity, ecosystems (nature) and nature's contributions to people and when appropriate for restoring nature. The Assessment is structured around the different subregions (North America, Mesoamerica, the Caribbean, and South America), taking into account the distinct biophysical features of major biomes (Intergovernmental Panel on Biodiversity and Ecosystem Services units of analysis) in each subregion and the multiple types of social and economic distributions of wealth and access to nature's contributions to people.

12. In this Assessment, we synthesize existing knowledge to quantify, to the extent possible, the magnitude and trends in nature's contributions to people enjoyed by the people of the Americas and assess how these contributions add to quality of life of various cultures in the region. We also assess the impact of several ongoing pressures on nature and nature's contributions to people including urbanization and depopulation of rural areas, natural resource exploitation, pollution, climate change, loss and degradation of natural habitats (terrestrial, freshwater, coastal and marine). Within subregions, these syntheses and assessments are done by major biomes with attention to socio-economic and cultural differences.

13. Our purpose is to make policy-relevant knowledge accessible and useful, working towards improved governance of and the sustainable use of nature and nature's contributions to people. To do this, we take a multidisciplinary and multi-knowledge systems approach. We identify the specific needs of each of the main American subregions regarding access to decision-support tools at different scales, knowledge gaps and capacity-building needs, including the development of capacity for future sustainable uses of nature and nature's contributions to people.

14. This chapter also introduces key concepts such as nature's contributions to people, units of analysis and the Intergovernmental Panel on Biodiversity and Ecosystem Services conceptual framework used in this Regional Assessment. Furthermore, this chapter introduces the key core questions posed by policymakers during the scoping phase prior to this Assessment and how several chapters in this Assessment address them. The target audience of this Assessment is primarily policymakers whose work may affect or be affected by nature or nature's contributions at all levels and the United Nations programmes and multilateral environmental agreements that are key clients for Intergovernmental Panel on Biodiversity and Ecosystem Services reports. A broader audience includes intergovernmental and non-governmental organizations, business and industry, practitioners, indigenous and local knowledge holders, community-based organizations, the scientific community, and the general public.

1.1 Overview of the region

The Americas covers the widest range of latitude of any of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) Regional Assessments. It includes wide expanses of deserts, grasslands, savannas and forests in different climatic conditions (polar, temperate, mediterranean, arid, sub-tropical, tropical) and topographic settings (plains, plateau, mountains). This region has the largest proportion of freshwater resources (Great Lakes and Amazon basin) and extent of rainforest, and the longest terrestrial mountain range (Andes).

The Americas include 55 of the 195 terrestrial and freshwater world ecoregions with highly distinctive or irreplaceable species composition (Olson & Dinerstein, 2002). The region hosts 20% of globally identified key biodiversity areas, 26% of globally-identified terrestrial biodiversity hotspots, and the Gulf of California and Western Caribbean are included in the top 18 marine biodiversity hotspots and conservation priorities for tropical reefs (Olson et al., 2001; Roberts et al., 2002; Marchese, 2015; UN, 2016a; World Database of Key Biodiversity Areas, n.d.). The region has some of the most extensive wilderness areas in the planet, such as the Pacific Northwest, the Amazon and Patagonia. It also contains the Mesoamerican reef, which is the largest barrier reef in the western hemisphere, and three of the six longest coral reefs in the world (World Atlas, 2017; WWF, 2017). The region is also a main center of origin and domestication for important crops such as potato, quinoa, maize, beans, cacao, tomatoes, squash, chili (Clement et al., 2010; Galluzzi et al., 2010; Parra & Casas, 2016). The Americas are home to globally outstanding terrestrial, freshwater and marine biodiversity, many of the richest biomes, and some of the world's most important biodiversity hotspots (e.g. Tropical Andes, Brazilian Cerrado and South American Atlantic Forest, California Floristic Province, Mesoamerica, Central Chile, western Ecuador, coral reefs of the Caribbean) (Myers et al., 2000; UN, 2016a). Well-functioning terrestrial, marine and freshwater ecosystems in the Americas underpin regulating functions highly relevant to environmental processes. These include functions such as the regulation of freshwater quantity, flow, and quality (Russi et al., 2013; Grizzetti et al., 2016), carbon and nutrient cycling (Anderson-Teixeira et al., 2012), moderation of extreme natural hazards (e.g. vegetation and wetlands help prevent floods), and coastal protection (coastal wetlands and coral reefs provide buffer against waves, storms, and sea level rise) (Ferrario et al., 2014; Van Zanten et al., 2014).

People's quality of life in the Americas is highly dependent on nature's material contributions (including food and feed, medicine, energy, fibers, and construction materials) to achieve food, water and energy security, and to generate income and support livelihoods and health. The region has the top producers of many agricultural commodities, such as sugar, coffee, and orange juice (Brazil) and coarse grains (USA) (The Economist, 2017). While the region has only 15% of the world's population, it accounts for 34% of the global Gross Domestic Product at purchasing power parity (GDP_{PPP}) in 2016 (UNDP, 2016; section 4.3.2), contributes around 41% of global ecosystems' biocapacity[1], and 23% of the world's ecological footprint (with 171% higher per capita ecological footprint than the global average) (Global Footprint Network, 2016).

The region is a mosaic of peoples living in diverse socio-economic and political settings with different values, world visions, and interests in nature and its benefits to them. The region still has large local populations producing cash and various subsistence products on small holdings or through small-scale fishing, with a considerable contribution to their local communities and nearby cities.

A good quality of life in the Americas is also based on non-material nature's contributions to people (NCP). Nature can help societies achieve a compassionate and equitable life and provide learning and inspiration for culture, identity, and social cohesion. The beauty of nature reflected in art and architecture has inspired communities and individuals for centuries. Some worldviews, especially from indigenous communities in the Americas (accounting for 5% of the population in the continent), show remarkable symbolic links with nature, some perceiving it as an entity with its own rights. For example, Bolivia and Ecuador explicitly recognize the importance of "Mother Earth and living in harmony with nature" in their legal frameworks

[1] In this assessment "biocapacity" is defined by the Global Footprint Network as "the ecosystem's capacity to produce biological materials used by people and to absorb waste material generated by humans, under current management schemes and extraction technologies". The "biocapacity" indicator used in the present report is based on the Global Footprint Network, unless otherwise specified.

(Gregor Barié, 2014; Guardiola & García-Quero, 2014; Pacheco, 2014). Several national parks and areas of biological significance have been created at sites of former sacred areas, for example the Alto Fragua Indiwasi National Park, the first Colombian national park, created at the request of indigenous communities, and the biodiversity reserve of the Wemindji Cree of James Bay in Canada (Pilgrim & Pretty, 2010). In addition to the importance of nature´s contributions for social cohesion, bonds and culture, several studies show positive linkages between healthy environments and healthy people**.** One example is the positive psychological benefits of green space and natural elements to people's satisfaction and well-being (Fuller et al., 2007).

Despite the importance of non-material contributions of nature to indigenous and local populations, decisions of land ownership and other rights to use and access resources have not been inclusive and evenly distributed among the diversity of inhabitants. However, there are institutional arrangements emerging across the region that are attempting to accommodate the plurality of values and interests. Some new arrangements include decentralization of rights to local communities to govern their natural resources, co-management between the state and private or local communities, and other mixes of arrangements among social actors.

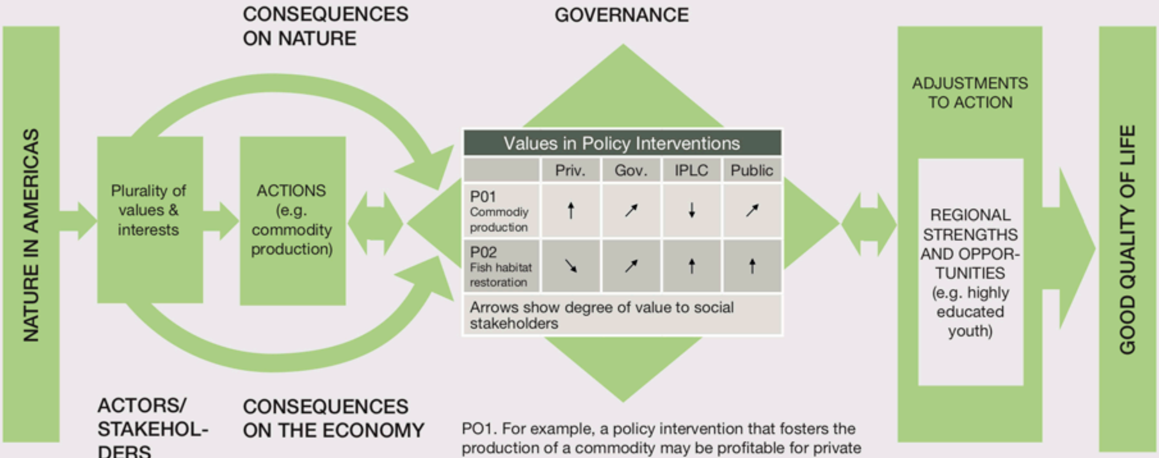
Given the nature of environmental problems that have no geographic boundaries, multi-boundary policies and cooperation are needed. Some examples include the subregional management of flying fish among eastern Caribbean countries (CRFM, 2014). An example of increasing regional cooperation between Argentina, Bolivia, Brazil, Paraguay and Uruguay to manage multi-boundary water resources is showcased in the Río de la Plata basin (Leb, 2015; Siegel, 2017). Another transboundary agreement, governance model, and cooperative initiative to manage water quality is found in the Great Lakes basin between the USA and Canada (Clamen & Macfarlane, 2015; Jetoo et al., 2015; Johns, 2017).

The diversity of the region, and nature's contributions to people are affected by policies, incentives, disincentives, and other decisions at all scales by altering positively and negatively how governments, institutions, and individuals interact with nature. Moreover, socio-environmental challenges are often shared between countries, which suggest that regional and subregional cooperation may be essential to find and enhance solutions (Gander, 2014). Because of these complexities, an integrated assessment of biodiversity and NCP is necessary to untangle the many interlinkages, at regional and subregional scales (Figure 1.1).

Figure 1 1 **The plurality of values and interests shaping governance processes and policy and decision-making in the Americas. Source: Own representation**

This figure illustrates two hypothetical cases of how a resource management decision flows through the dynamics of governance. Typically, diverse values and interests of people will inherently have trade-offs, with choices benefiting some while costing others, and with consequences for nature and the economy. Governance is where and how choices on the use of nature are made, depending on actors' values and interests.

Policy interventions that take into account these economic and environmental consequences and take advantage of regional strengths as opportunities (such as the large social capital, institutional diversity, widespread endorsement of international environmental agreements) are showing greater potential to achieve an inclusive sustainable development and better quality of life in the Americas.



ACTORS/ STAKEHOL-DERS

Governments, Public, Private Indigenous peoples and local communities (IPLC)

PO1. For example, a policy intervention that fosters the production of a commodity may be profitable for private sector actors but may result in the loss of high value land to IPLC groups as ancestral spiritual places. Central governments and some businesses may assign a high value to the economic dimension of this intervention (e.g. tax revenue), while IPLC groups may feel that their quality of life would be harmed by the same intervention.

PO2. In an area with overfishing and vulnerable IPLC, a policy that fosters fish and habitat restoration, including access restriction, may improve local income and food security for IPLC and other stakeholders when employment and recreational options are increased. Commercial fisheries may however reduce profits due to limited access to this area.

1.2 The core policy-relevant questions for the Americas Assessment

Given the complexity of environmental problems and processes, decision makers in civil society, governments and the private sector have expressed their need for IPBES experts to answer key core questions specific for the American continent. These requests and suggestions put forward by governments, stakeholders and multilateral environmental agreements were submitted to IPBES. Experts, selected by the IPBES Multidisciplinary Expert Panel (MEP), assessed the scope of Regional Assessments and reached consensus on the contents to be included in each chapter of the Assessment. The resulting scoping assessment was approved by the IPBES Plenary in 2015 and was the foundation for developing this Regional Assessment for the Americas (IPBES, 2015a). Consequently, the Americas Regional Assessment is expected to address the following policy-relevant questions:

(a) How do biodiversity and ecosystem functions and services contribute to the economy, livelihoods, food security, and good quality of life in the region, and what are their interlinkages?

(b) What are the status and trends of biodiversity and ecosystem functions underpinning NCP that ultimately affect their contribution to the economy, livelihoods and well-being in the region?

(c) What are the pressures driving the change in the status and trends of biodiversity, ecosystem functions, ecosystem services and good quality of life in the region?

(d) What are the actual and potential impacts of various policies and interventions on the contribution of biodiversity, ecosystem functions and ecosystem services to the sustainability of the economy, livelihoods, food security and quality of life in the region?

(e) What gaps in knowledge need to be addressed in order to better understand the distribution of biodiversity and assess drivers, impacts and responses of biodiversity, ecosystem functions and services at the regional and biome levels?

The Americas Regional and Subregional Assessment on biodiversity, ecosystem function and ecosystem services is designed to provide a credible, legitimate, holistic, and comprehensive analysis of the current state of scientific and other types of knowledge. It will analyze options and policy support tools for sustainable management of biodiversity, ecosystem function and ecosystem services under alternative scenarios and present success stories, best practices, and lessons learned, including progress made in the Strategic Plan for Biodiversity 2011–2020 and its Aichi biodiversity targets, the Sustainable Development Goals (SDGs), and the National Biodiversity Strategies and Action Plans developed under the Convention on Biological Diversity (CBD). This Assessment will also identify current gaps in capacity and knowledge and options for addressing them at relevant levels.

1.2.1 How do biodiversity and ecosystem functions and services contribute to the economy, livelihoods, food security, and good quality of life in the region, and what are their interlinkages?

Nature (biodiversity and ecosystems) and the contributions it makes available to people (ecosystem functions and services) referred as NCP are essential to achieve a good quality of life in the Americas. Economies and societies depend –to different extents– on NCP to achieve food, water and energy security, generate income and support livelihoods and health. This includes food and feed, medicine, energy, fibers, and construction materials. Nature´s regulating contributions are critical for environmental functions such as the regulation of freshwater quantity, flow and quality (Kimbell & Brown, 2009; Mueller et al., 2013; Russi et al., 2013; Grizzetti et al., 2016). These contributions are essential to foster water security[2] in the Americas (see Chapter 2). They can be threatened by climate change and by excessive extractive uses affecting mainly water users, business operations, and biodiversity (Postel, 2000; Ramsar, 2008; Gleeson et al., 2012).

A good quality of life, shaped by one's worldview, can be interpreted as how non-material nature's contributions help societies achieve a compassionate and equitable life, and provide learning and inspiration for culture, identity, social cohesion and symbolic bonds with nature. It can also encompass the relationships between humans, land, plants, animals, mountains and other sacred elements (Chapter 3).

There is considerable evidence of the harmful effects of nature's degradation on public health, livelihoods and both regional and national economies in the Americas (see Chapters 2-4). Pollution is considered the number one cause of death and disease, contributing to an estimated nine million premature deaths (Das & Horton, 2017). Harmful effects of environmental degradation (e.g. air pollution, land degradation, natural disasters) disproportionately affect the poorest populations and therefore pose a threat to inclusive development (WB & IHME, 2016). Often, the poorest segments of societies live and work in polluted environments and are most vulnerable to natural disasters and the impacts of extreme weather events, which leads to increasing inequality (Scarano & Ceotto, 2015; Young et al., 2015). Industrial facilities and other sources of air pollution have often been sited close to poor minority communities, which lead to inequitable exposure to poor quality environments (Morello-Frosch et al., 2011). In poor urban neighborhoods, asthma rates are far greater than the national average (Claudio et al., 2006).

Recent decades have seen the development of research at the interface of ecology, economics (e.g. TEEB, 2010; Haines-Young & Potschin, 2012) and human demographics (Aide & Grau, 2004) that describe the

[2] In this assessment "water security" is used to mean the ability to access sufficient quantities of clean water to maintain adequate standards of food and goods production, sanitation and health care and for preserving ecosystems

complex interdependence of NCP, economies and well-being. These studies focus on drivers of change in land use and patterns of biodiversity and potential outcomes for NCP and human well-being. For example, agricultural lands are the world's largest managed ecosystem, now covering 40% of global terrestrial surface (Foley et al., 2005). The changes of vegetation were made to enhance a single provisioning service – food for people (Wood & DeClerck, 2015), but this has come at the cost of significant degradation of water quality and quantity, increased greenhouse gas emissions, disruption of natural pest control, pollination and nutrient cycling processes (Matson et al., 1997; Diaz & Rosenberg, 2008; Klein et al., 2009) and has impacted the livelihoods of local and Indigenous Peoples tied to their natural environments (Altieri & Toledo, 2011; DESA, 2014). However, current research indicates that agricultural lands can become significant providers of many ecosystem services, depending on their design and management (Kremen & Miles, 2012; Zhang et al., 2014; Wood & DeClerck, 2015) as well as on function and the diversity within and the surrounding landscape (Kremen & Ostfeld, 2005; MEA, 2005; Tscharntke et al., 2005; TEEB, 2010).

Exploring this issue contributes to understanding relationships among economy, livelihoods and well-being in the region. Finding solutions will require integration across social and ecological systems and investigation of questions about how ecosystem services are co-produced by social systems of management and ecosystem design; how costs and benefits from alternative approaches of NCP use are distributed among sectors of societies and cultures, and consequences of alternative practices for governance of nature and its uses (Bennett et al., 2015). The Assessment will also explore how today's answers to the questions may shift in response to major drivers, including climate change (e.g. FAO, 2013), cultural preferences, and shifting patterns of land use. The Americas is the most urbanized region worldwide (UN, 2013). In the last five decades, the proportion of the population of Latin America and Caribbean living in rural areas has dropped significantly, as populations become concentrated in urban centers (DESA, 2014). Perhaps most importantly the Assessment will review situations traditionally presented as requiring direct trade-offs among pairs of alternative uses of specific NCP in broader conceptual terms, considering the full range of NCP collectively, the distribution of benefits and costs among the full range of people affected by the trade-offs, and the multiplicity of worldviews about the values attached to the different NCP.

1.2.2 What are the status, trends of biodiversity and ecosystem functions underpinning nature's benefit to people that ultimately affect their contribution to the economy, livelihoods and well-being in the region?

The status and trends of biodiversity and NCP cannot be interpreted independent of the policy framework in which the Assessment is conducted. To illustrate, increases in food production and exports may be seen by policy makers as progress towards their specific goal to increase quality of life of the poor by intensifying use of nature's contributions (e.g. the 10 year projections of agriculture output of the Brazilian agricultural research centre and Argentina's, Colombia's and others in the Amazon basin). However, although intensification of agriculture can increase GDP or Human Development Index (HDI), if not done sustainably, it can lead to loss of ecosystems and their services (FAO, 2013; Venter et al., 2016) that can have downstream affects. The loss of feeding and reproduction habitats in floodplains of the Amazon due to conversion to agriculture could dramatically affect fisheries in the Amazon Delta, which is one of the pillars of traditional and industrial economies there. Consequently, in this Regional Assessment, the status and trends in terms of impacts on biodiversity, extinction rates, and ecosystem health are assessed. Any documented trends, and status relative to descriptive benchmarks (like average for the past decade) may then be interpreted relative to a various goals governments and sectors of society may have for the area.

Throughout this Assessment we refer in some places to nature, and in other places to biodiversity. When reference is made to "nature" the intent is to refer to nature in a holistic and unified way – all its structural components, its functional relationships and processes, and the place of humanity within it. When the Assessment is considering specific pieces of nature – populations species, communities or ecosystems, the component functions and processes, or human uses of or impacts on specific aspects of nature, the term biodiversity will be used. The associated text will often include adjectives or phrases to make clear what scale and aspect of "biodiversity" is being discussed.

1.2.3 What are the pressures driving the change in the status and trends of biodiversity, ecosystem functions, ecosystem services and good quality of life in the region?

In the IPBES conceptual framework guiding this Assessment (Diaz et al., 2015), drivers of change refer to all those external factors that affect nature, anthropogenic assets, nature´s contribution to people and a good quality of life. Drivers of change include institutions and governance systems and other indirect drivers, and direct drivers both natural and anthropogenic. Quantifying to the extent possible the magnitudes and trajectories of the drivers in the IPBES framework is an important step in the Regional Assessments, but using that information requires taking into account how drivers interact with nature, NCP, economies, societies and cultures, and with each other.

Consideration of these interactions is at the heart of the IPBES Assessment. In any landscape or region, there is a diversity of social actors who utilize the same landscapes and resource base. To illustrate, there is diversity in livelihood strategies across the Amazon. If economic drivers provide incentive to create infrastructure needed extract the specific goods desired by the markets, there will be diverse responses. Greater wealth from the enhanced trade may increase in price and demand of goods locally as well, to which local populations/smallholders and large-holders may respond differently. The differential responses then affect the ability of the land and coastline to provide other NCP (fish habitat, water regulation), with potential additional conflicts between groups and encroachment on indigenous lands and smallholder areas, and the infrastructure may change the many non-material NCP. The Assessment gives importance to tracking such linkages and interdependencies among drivers.

1.2.4 What are the actual and potential impacts of various policies and interventions on the contribution of biodiversity, ecosystem functions and ecosystem services to the sustainability of the economy, livelihoods, food security and good quality of life in the region?

Different policies and interventions related to biodiversity, ecosystem functions and services are contributing to a good quality of life for peoples in the Americas, which include achieving food security, and supporting livelihoods and public health as well as the sustainable development of local and regional economies.

Policies affecting nature and NCP include a wide array of tools and practices that address on one side, the conservation and restoration of nature and on the other side, the management of impacts of development on nature. In the Americas, policy tools that are designed to conserve nature include protected areas, ecological or biological corridors, indigenous and community conserved areas, and conservation incentives such as payment for ecosystem services, eco-certification and sustainable investments. Other policy tools seek to reduce the impact of development on nature by regulating the extent and ways that development can alter nature, used enablers such as environmental impact assessments, which are intended to evaluate the environmental consequences of a development activity or project before implementation. Around the Americas many combinations of these policy strategies and tools are used, according to the capacities, legislation, traditions and values of the specific area.

The Americas region has had many successful experiences in biodiversity conservation, restoration and sustainable use at regional and local levels and in terrestrial, freshwater, coastal and marine systems, as well as failures to keep uses sustainable (UNEP, 2012; Bennett et al., 2017). The resulting lessons learned need to be assessed and understood to inform the development of appropriate policies that ensure sustainability (Foley et al., 2011). However, future policies will function in a context of climate change, teleconnections to other regions, population growth, industrialization and development, and the consequent changes in demand for food, water, biomass, and energy. Consequently, past policies to address these types of pressures and demands need to be periodically re-evaluated in the context of these changes in pressures (Foley et al., 2005). In some cases, the magnitude of these impacts on biodiversity and ecosystems are thought to threaten economies, livelihoods and quality of life (IPBES, 2014). However, even the nature of an individual threat can vary among sectors of society, depending both on culturally based views of the value of biodiversity and specific ecosystem services and how the benefits and impacts associated with the uses of the NCP are distributed.

A vast array of such policies have been assessed in the Americas, including conservation incentives (e.g. watershed protection initiatives), protected areas, indigenous and community conserved areas, ecosystem restoration, eco-certification and investments that account for environmental, social and governance factors in portfolio selection and management. In most cases, there were some unexpected or undesired results, indicating that the breadth and depth of planning for use of these instruments has scope to improve (Wuenscher et al., 2008; Engel et al., 2008; Joppa & Pfaff, 2009; Arriagada et al., 2012; Miteva et al., 2012; Watson et al., 2014; Barral et al., 2015; Ferraro et al., 2015; Baylis et al., 2016; Juffe-Bignoli et al., 2016; Rodríguez Osuna et al., 2017; Vörösmarty et al., 2018).

The impact of different interventions and policies vary widely across the Americas and are often a result of a combination of more than one intervention. For example, a decline in deforestation in Brazil in the past decade was the result of the combined effect of: (a) public and private partnership (b) the banning of soybeans and beef produced in deforested lands (c) improved monitoring and enforcement to combat deforestation, and (d) the 2008 global financial crisis on commodity demand (Nepstad et al., 2014; Cisneros et al., 2015). Separating the effect of single components is complex and case specific (Nepstad et al., 2014).

The effectiveness and impact of policies and interventions related to nature's components depend on the way societies perceive the world, negotiate interests, prioritize problems, and find feasible solutions that respect social, institutional, and environmental settings. Such enabling conditions are essential to foster a successful implementation of policies that include environmental and other societal issues (e.g. poverty reduction, including local knowledges and minorities). Current international policy strategies, goals and high level commitments for the protection of nature and sustainable development are driving changes in the same direction and thus creating synergies (UN, 2015; Dicks et al., 2016; UN, 2016b).

1.3 Background to the Intergovernmental Platform on Biodiversity and Ecosystem Services Regional Assessments

Assessments that examine the relationships between policy goals and ecosystem services can inform decision makers whose goals and actions are focused on people, society, and economies (Ash et al., 2010). The Millennium Ecosystem Assessment (MEA) concluded that the provision of the majority of ecosystem services is declining and their availability into the future cannot be taken for granted. It also concluded that the failure to consistently give adequate weight to the dependence of human well-being on biodiversity and ecosystems in public and private decision making has allowed those services to be degraded, increasingly compromising our ability to achieve long-term development goals (MEA, 2005). The concept of ecosystem services gained prominence in the MEA (2005). In the years since the MEA, the term ecosystem services has been taken up by many disciplines and user groups, including Environmental Economics, Integrated Ecosystem Assessments, and Spatial Planning (both terrestrial and marine). As the interest in and uses of ecosystem services has increased, interpretation of the term has evolved and diversified (Chan et al., 2016; Gomez-Baggethun et al., 2016). Some uses, particularly in environmental economics, have been interpreted as de-emphasising the ecosystem services that are not readily monetized for use in commerce and optimization or trade-off analyses (e.g. TEEB, 2009).

The Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services[3] was established to strengthen the science–policy interface for the conservation and sustainable use of biodiversity, ecosystem services, long-term human well-being, and sustainable development. It has similarities to the

[3] www.ipbes.net

Intergovernmental Panel on Climate Change in that both carry out assessments using existing knowledge to addresses questions where the knowledge bases are complex, incomplete and uncertain, making straightforward answers not possible (Perrings et al., 2011). In addition, although the biodiversity crisis is global, biodiversity distribution and its conservation status are heterogeneous across the planet. Consequently, governments and other stakeholders require information for solutions that are scalable to multiple levels (Diaz et al., 2015).

In this context, the IPBES agreed to conduct four Regional Assessments for the Americas (including the Caribbean islands); Africa; Europe and Central Asia; and Asia and the Pacific. The Americas region comprises a land area of 39 million square km, extending from Arctic to sub Antarctic latitudes. The Americas include some of the most biodiverse biomes in the world (Olson et al., 2001). The Americas region is also culturally diverse with some of the most industrialized urban areas on the planet. This poses a challenge to find ways for different cultures to co-exist and share these ecosystems (Kipuri, 2009). However, it also presents opportunities such as the chance to draw upon the traditional knowledge of indigenous people and local communities when conducting IPBES Assessments. The tensions between these challenges and opportunities from cultural diversity and the different knowledge systems pervade the IPBES Assessments.

1.3.1 What are nature contributions to people?

The Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services considers NCP as an inclusive set of categories across knowledge systems, which comprise all nature's contributions, both positive and negative, to human quality of life. People obtain these contributions entirely from nature or, more often, apply knowledge and work to co-produce benefits with nature (Pascual et al., 2017). Concerns over the potential for misinterpretation of the categories of ecosystem services led IPBES to use NCP instead of ecosystem services, to ensure cultural and aesthetic relationships between people and nature are considered on an equal plane with other ways that people relate to and use nature. Additionally, some feel the new term may aid with the integration of multiple disciplines and answering of policy-relevant questions that are central to the IPBES mission. IPBES utilizes the term "good quality of life" instead of "well-being", which is conceived to comprise aspects such as access to food, water, energy and livelihood security but also human health, equitable social relationships, cultural identity, and freedom of choice and action (Pascual et al., 2017).

There are many categorizations of NCP, which evolved from the concept of ecosystem services (MEA, 2005). The Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services has decided to use a set of 18 NCP, distinguishing three broad groups - regulating, material, and non-material contributions. Regulating contributions are functional and structural aspects of organisms and ecosystems that may modify environmental conditions experienced by people, or sustain or regulate the generation of material and non-material benefits. In many cases, these regulating NCP are not perceived directly, but nevertheless may be essential to life. Material contributions are substances, objects or other material elements taken from nature that help to sustain people's physical existence and infrastructure. They are typically consumed and consciously perceived as food, energy, or materials for shelter or ornamental purposes. Non-material contributions affect people's subjective or psychological quality of life, individually and collectively. The entities may be physically consumed or altered in the process (e.g. animals in recreational or ritual fishing or hunting) or not (individual biodiversity components or ecosystems as source of inspiration).

The 18 categories of NCP are listed in Table 1.1. Collectively, these categories include all potential ways that nature contributes to human quality of life. As developed in Chapter 2, many of these NCP are essential to achieve a good quality of life in all cultures, whereas the values attached to others, especially some material and non-material contributions, can be influenced strongly by one's culture, economic status, and worldview (IPBES, 2015b). The use of these standardized categories of NCP brings a common structure to all the Regional Assessments, but presents challenges when referring to literature that uses other classifications and terms. This is particularly challenging for using the more recently adopted term NCP rather than ecosystem services.

Table 1.1. The NCP used by IPBES for linking human well-being and nature

Regulating Contributions

- Habitat creation and maintenance – maintaining the ecosystem structures and processes that allow the other NCP to be provided
- Pollination and dispersal of seeds and other propagules – the ways that nature contributes to productivity of plants through fertilizing seeds and dispersing seeds and other vegetative propagules (IPBES, 2016a).
- Regulation of air quality – regulation of CO_2/O_2 balance, Ozone for ultraviolet-B absorption, polluting gases
- Regulation of climate – including regulating albedo, some aspects of greenhouse gas emissions, and carbon sequestration
- Regulation of ocean acidification – maintaining the pH of the ocean through buffering the increases and decreases of carbonic acid (caused mainly by uptake of atmospheric carbon dioxide in the oceans)
- Regulation of freshwater quantity, location and timing – for both direct uses by people and indirectly for use by biodiversity and natural habitats
- Regulation of freshwater and coastal water quality – capacity of healthy terrestrial and aquatic ecosystems to regulate water supply delivery and/or filter, retain nutrients, sediments and pathogens affecting water quality
- Formation, protection and decontamination of soils and sediments – sediment retention and erosion control, soil formation and maintenance of soil structure, decomposition and nutrient cycling
- Regulation of natural hazards and extreme events – preserved ecosystems' role in moderating the impact of floods, storms, landslides, droughts, heat waves, and fire
- Regulation of organisms detrimental to humans – pests, pathogens, predators, competitors

Material contributions

- Energy – biomass-based fuels
- Food and feed – wild and domesticated sources, feed for livestock and cultured fish
- Materials and assistance – production of materials derived from organisms in crops or wild ecosystems, for construction, clothing, printing, ornamental purposes and decoration
- Medicinal, biochemical and genetic resources – plants, animals and microorganisms that can be used to maintain or protect human health directly or through process of the organisms or their parts

Non-material contributions

- Learning and inspiration – opportunities from nature for the development of the capabilities that allow humans to prosper through education, acquisition of knowledge and development of skills
- Physical and psychological experiences – opportunities for physically and psychologically beneficial activities, healing, relaxation, recreation, leisure, tourism and aesthetic enjoyment
- Supporting identities - basis for religious, spiritual, and social-cohesion experiences, for narrative and story-telling and for sense of place, purpose, belonging, rootedness or connectedness
- Maintenance of options – continued existence of a wide variety of species, populations and genotypes, to allow yet unknown discoveries and unanticipated uses of nature, and on-going evolution

Source: IPBES (2017a)

Nature Contributions to People will be the term used in all Chapter summaries and in the Summary for Policy Makers, to make sure this broad meaning is communicated unambiguously. However, when summarizing the information used sources taken from publications, particularly the information from scientific sources, those sources frequently use "ecosystem services" in ways specific to the discipline of the author. To substitute NCP in those cases would sometimes misrepresent the meaning intended by the sources. Consequently, in the body of the Chapters of this Assessment, "ecosystem services" will be used and the context explained, as necessary, to present the information from the source accurately.

1.3.2 Why are nature contributions to people relevant to human quality of life (well-being and livelihoods) in the Americas?

Human's quality of life in the Americas is highly dependent on Nature's material contributions to achieve food and energy security, generate income and support livelihoods and health. This includes food and feed, medicine, energy, fibers, and construction materials (Chapter 2). In terms of food, the Americas is an important commodity producer, where Brazil, USA, Mexico, Canada, Honduras, Peru, Argentina, Ecuador, Dominican Republic, Colombia and Guatemala are amongst the top 10 producers of commodities, including wheat, rice, sugar, coarse grains, tea, coffee, cocoa, and orange juice. Brazil is the top producer of sugar, coffee and orange juice. The USA is the most important producer of coarse grains, which include corn, barley, sorghum, oats, rye, millet, triticale and others. Six countries in the Americas have the largest agricultural output in terms of agriculture and fisheries: USA with $226 billion in 2013 and Brazil with $111billion in 2014 (The Economist, 2017). This region has also some of the biggest producers of cereals, meat, fruit, vegetables, roots and tubers, as well as fisheries and aquaculture products (USA, Brazil, Mexico, Argentina). In terms of biomass-based fuels, the USA, Brazil and Argentina are the world top three major oil seed (soybeans, rapeseed, cottonseed, sunflower seed and groundnuts) producers (The Economist, 2017). Food production (including commodities) contributes positively to some aspects of human well-being (short and medium-term GDP) but it can also generate a series of environmental externalities (in the short, medium and long-term) that have negative effects on nature and people. Some examples include pollution derived from fertilizer application (nutrient runoff) from agricultural sites into freshwater systems, which result in harmful impacts on freshwater resources, biodiversity, air quality, and coastal systems (Mekonnen & Hoekstra, 2015; Chapter 4).

Medicines provided from nature have been used for several thousands of years to treat disease and injuries, and relieve pain. Despite rapid progress in drug development, most prescribed medicines used in developed countries are still based on or patterned after natural compounds found in animals, plants and microbes (Chivian & Bernstein, 2010). This is especially relevant for drugs designed to treat infections and cancer. Other examples include aspirin from the White Willow Tree (*Salix alba vulgaris*), morphine from the Opium poppy (*Papaver somniferum*); azidothymidine used to treat HIV/AIDS (Human Immunodeficiency Virus Infection / Acquired Immune Deficiency Syndrome) patterned after marine sponge compounds *Cryptotethya crypta* (Chivian & Bernstein, 2010). Diets based on natural products and active livelihoods of indigenous groups (Tsimane) in the Bolivian Amazon is an example of significantly positive health outcomes (lowest reported levels of coronary artery disease of any population to date) (Kaplan et al., 2017).

A good quality of life in the Americas is also based on nature's non-material contributions, which help societies achieve a compassionate and equitable life and provide opportunities for learning and inspiration for culture, identity, social cohesion and symbolic bonds with nature (IPBES, 2017a). The beauty of nature reflected in art and architecture has inspired communities and individuals for centuries. Some worldviews especially from indigenous communities in the Americas show remarkable symbolic links with nature, perceiving it as an entity with own rights. For example, some South American countries (Bolivia and Ecuador) explicitly recognize the importance of "Mother Earth and living in harmony with nature" in their legal frameworks as means to provide a good quality of life (Gregor Barié, 2014; Guardiola & García-Quero, 2014; Pacheco, 2014; Estado Plurinacional de Bolivia, 2015). It is no coincidence that several national parks have been created at sites of former sacred natural areas, for example the Alto Fragua Indiwasi National Park, the first Colombian national park, created at the request of indigenous communities (Pilgrim & Pretty, 2010). Sacred natural areas recognized by UNESCO (United Nations Educational, Scientific and Cultural Organization) in the Americas include the Gran Ruta Inca, the ancient route across the Andean highlands, American indian sacred springs and waters of New Mexico, Sacred sites and gathering grounds initiative in Arizona, Sacred lakes and springs, Huascarán world heritage site and Biosphere Reserve in Peru (Schaaf & Lee, 2006). Similarly, in Canada, a biodiversity reserve was established at the request of an indigenous group, the Wemindji Cree of James Bay (Pilgrim & Pretty, 2010). Non-material contributions have served functions cross-culturally as well as within cultures. For example, aquatic ecosystems have historically been a means for promoting cooperation and resolving conflict, and thus serve an important societal role, mainly for international, transboundary watersheds (UNEP-DHI & UNEP, 2016).

In addition to the importance of nature´s contributions for social cohesion, bonds and culture, studies show positive linkages between healthy environments and healthy people (Maller et al., 2006). One example is the positive psychological benefits of greenspace and natural elements to people's satisfaction and well-being (Fuller et al., 2007; Kaplan et al. 2017).

Nature in the Americas also underpins regulating functions (regulating contributions) highly relevant to environmental processes that are essential to people's good quality of life such as the regulation of freshwater quantity, flow and quality. Forests and wetlands are the ecosystems mostly recognized for their role in the regulation of freshwater supplies, which is abundant in the region (compared to the global average) but unevenly available across geographies and time (Green et al., 2015). Some cities in South America face severe water scarcity episodes during specific times of the year (Bogotá, Quito, La Paz, Lima) as well as in USA states such as California, Texas and Florida. Areas with high scarcity occur where densely populated areas compete for water with intensely irrigated agriculture, or areas with reduced water storage (Buytaert & De Bièvre, 2012; Mekonnen & Hoekstra, 2016).

The importance of such regulating contributions is showcased by the now-classic example of the city of New York paying for upstream watershed protection rather than investing in constructing more expensive additional filtration plants (Hanson et al., 2011; McDonald et al., 2016). These types of contributions are essential to foster water security as well as other benefits in the Americas (Ramsar, 2008; WWAP, 2015). Conserved areas are key to providing with drinking water for several important cities of the Americas including in the USA, Brazil, Colombia and Venezuela (WB & WWF, 2003; Pabon-Zamora et al., 2008; Dudley et al., 2016; Harrison et al., 2016; Hermoso et al., 2016). However, choices of this type also illustrate the complexity of such policies; upstream watershed protection measures require residents and traditional users associated with the protected forests to accept financial payments in exchange for constrains on development opportunities and possibly some traditional forest uses, far from the urban area where the water is used. Other contributions of nature to regulate freshwater quality are related to wetlands that deliver well-documented benefits in waste treatment (e.g. wetlands and other aquatic ecosystems remove waste, recycle nutrients and dilute pollutants) and thereby act as natural water purification plants (De Groot et al., 2002; Russi et al., 2013; Zhang et al., 2014; McDonald et al., 2016). The flows of freshwater ecosystems are also important for energy production (for example, most of electricity generation in the USA comes from power plants that rely on water resources for cooling), which can affect power output and reliability (Feeley et al., 2008; Macknick et al., 2011; EIA, 2017).

Other important regulation functions that nature provides include the regulation of climate hazards and extreme events. Vegetation reduces the impact and likelihood of snow avalanches and landslides and coastal wetlands can moderate floods (Hawley et al., 2012). For example, social and economic losses as a result of extreme weather events in Brazil (i.e. flooding, flash-floods and landslides) between 2002-2012 have caused significant damage valued between $57.21 to 113.1 billion or 0.4 to 0.9% of Brazil's accumulated GDP in that period (Young et al., 2015). The state of Rio de Janeiro reported that 45% of all national deaths were associated with such hazards (Young et al., 2015). In the USA, six climate-related hazards resulted in health and social costs in the order of $14 billion between 2000 and 2009 (Knowlton et al., 2011).

Box 1.1 Nature´s contributions to people in the Amazon

The Amazon region presents a high diversity of peoples' values and interests in how to use, interact and experience nature to guarantee a good quality of life. Nature in the Amazon has a wealth of ecosystems and biodiversity that are indispensable to delivering contributions to people NCP across scales (e.g. the Amazon river basin is one of the most mega-biodiverse and the largest source of freshwater in the world) (Marengo, 2006; Tundisi et al., 2015; Winemiller et al., 2016). At local scales, these benefits include those enjoyed as spiritual, social cohesion and cultural continuity as well as those managed as agricultural, mining, forestry, pharmaceutical and fishery commodities. For example, Amazon rivers and its seasonally flooded forests provide habitats for fish that support livelihoods of thousands of people (Tundisi et al., 2015). At landscape to regional scales, Amazon's forests regulate hydrological cycles (Veiga et al., 2004), water quality, and nutrient cycling that supports freshwater biodiversity and people (Menton et al., 2009). At continental to global scales, the importance of the Amazon in the regulation of the global carbon cycle is well recognized (Anderson-Teixeira et al., 2012; Pinho et al., 2014; Phillips & Brienen, 2017). This includes the forest's role in carbon sequestration (approx. 120 billion metric tons of C biomass), climate patterns (Pires & Costa, 2013; Tundisi et al., 2015) and extreme events such as floods and droughts (Nazareno & Laurance, 2015).

1.3.3 Why are people relevant to nature's ability to provide nature contributions to people?

The interaction between people and nature can affect nature's ability to provide regulating, material and non-material contributions; as illustrated in section 1.3.2. Policy decisions can enhance nature's ability to provide NCP, such as the upstream watershed protection example above. However, people's decisions can also contribute to nature's degradation, leading to negative impacts on health, livelihoods, regional and national economies, as well as other dimensions of good quality of life (MEA, 2005). The degradation of nature frequently involves the loss of natural assets (MEA, 2005; TEEB, 2009). Typically, these losses are not taken into account by traditional economic measures (TEEB, 2009; Costanza et al., 2014). The use of many traditional economic indicators often has resulted in a country depleting a natural resource base such as forests to provide positive gains measured by a specific valuation method such as GDP gain. Resource depletion has many other consequences that may affect people's quality of life, including the degradation of non-material contributions (recreation, spirituality, religion, and identity). This shortcoming has prompted interest in a broader range of more inclusive economic measures under way in international finance and development agencies (see Chapter 2).

This Regional Assessment confronts the complex links between nature's contributions to people and a good quality of life for the diverse cultures and worldviews in the Americas. Within Chapter 2, the Assessment first describes key nature's contributions to people for the subregions and major biomes in the Americas. In most of the Americas, multiple cultures share NCP, and the chapter also discusses the different values these cultures may associate with specific NCP. Based on key indicators, the status of those contributions is assessed. Subsequent chapters then develop the reciprocal interactions of people and nature, in the contact of how NCP contribute to and are affected by those interactions.

1.3.4 Why do we need a Regional Assessment?

Biodiversity, ecosystem functions and NCP make essential contributions to the economy, livelihoods and good quality of life of people throughout the world (CBD, 2010; UN, 2015; CBD/FAO/WB /UNEP/UNDP, 2016). The Strategic Plan for Biodiversity 2011–2020 and its Aichi Biodiversity targets seek to provide an overarching framework for effective and urgent action to manage biodiversity in order to ensure that by 2020 ecosystems are resilient and continue to provide essential functions and services, thereby contributing to peoples' quality of life and poverty eradication. These considerations are also included in the ongoing development of the Post-2015 UN (United Nations) Development Agenda and the associated SDG.

Regional and national biodiversity strategies and action plans are important vehicles for implementing the Aichi biodiversity targets and adapting them to regional and national conditions. Implementation strategies and plans are also being developed at multiple scales for the SDG. These strategies and action plans need to be informed about the linkages between NCP and good quality of life of diverse cultures and societies, in part because these linkages make the Aichi targets and SDG themselves interdependent. These interdependencies among these goals and targets provide opportunities to build on synergies, such as actions to protect upstream forests (for their role in regulating freshwater quality and their provision to downstream users) that directly contribute to achieve several goals: SDG 15 related to the protection and restoration of terrestrial ecosystems, SDG 6 on clean water and sanitation, SDG 11 on sustainable cities and communities, SDG 13 on climate action and SDG 3 on good health and well-being. However, planning must also take account of potential tensions among the SDG, such as efforts to promote SDG 14 on a healthy ocean must still find ways to allow harvesting of seafood to increase, as an essential contribution to SDG 2 on food security. Without the types of integrated assessments represented by IPBES, the development of policies and action plans for goals like the Aichi targets or SDG would not be informed of how to take these interactions into account. Moreover, assessments at regional and subregional scales are important, since these scales are ones where the synergies and tensions are often expressed and must be taken into account in policies.

Efforts to meet these targets thus require a strong knowledge base and strengthened interplay between scientists and policymakers, and between different knowledge systems to which the regional and subregional assessments are well placed to contribute (Griggs et al., 2013; Bhaduri et al., 2016).

1.3.5 What is an Intergovernmental Platform on Biodiversity and Ecosystem Services Regional Assessment?

An IPBES assessment is a critical evaluation of the state of knowledge in biodiversity and NCP. It is based on existing peer-reviewed literature, grey literature and other available knowledge such as indigenous and local knowledge. It does not involve the undertaking of original primary research. The Assessment involves a literature review (scientific articles, government reports, indigenous and local knowledge and other sources), but is not limited to such a review. The process of evaluating the state of knowledge involves the analysis, synthesis and critical judgement of information by more than 100 international experts from 23 countries over three years, and then aided by the assignment of clear confidence terms, the presentation of such findings to governments and relevant stakeholders on their request. IPBES Assessments focus on what is known, but also on what is currently uncertain. Assessments play an important role in guiding policy through identifying areas of broad scientific agreement as well as areas of scientific uncertainty that may need further knowledge generation such as through scientific research.

Regional Assessments are also a vehicle for the implementation of IPBES's functions, such as capacity building, the identification of knowledge gaps, knowledge generation, and the development of policy support tools. Furthermore, the Assessment is critical to furthering IPBES's operational principle of ensuring the full use of national, subregional and regional knowledge, as appropriate, including by ensuring a bottom-up approach (Schmeller & Bridgewater, 2016).

The Regional Assessments inform a range of stakeholders in the public and private sectors and civil society, including indigenous people and local communities, who will all benefit from sharing information and data that allows progress to be made towards the Aichi Biodiversity targets and the SDG. The Americas Assessment provides users with a credible, legitimate, authoritative, holistic and comprehensive analysis of the current state of biomes within regional and subregional biodiversity and ecosystem services and functions, based on scientific and other knowledge systems, and with options and policy support tools for the sustainable management of biodiversity and ecosystem services and functions under alternative scenarios; it also present success stories, best practices and lessons learned, identifying current gaps in capacity and knowledge and options for addressing them at relevant levels.

1.3.6 Who are the target audiences of this document?

Some primary and broader target audiences for IPBES's outputs are listed below although the list is not exhaustive, and many other categories of users may find the assessments useful in pursuing their mandates or goals:

1) Primary target audiences:

a) Policymakers whose work may affect or be affected by biodiversity, ecosystem services or NCP at all levels: IPBES member States, ministries of environment, energy, industry, planning, finance and agriculture, local authorities and the scientific advisers of policymakers need to be informed about IPBES so that they can use it as a source of independent expert knowledge;

b) b)UN programmes and multilateral environmental agreements: such as the CBD, and the Convention on Migratory Species, but also UN programmes with broad mandates for development and uses of planetary resources, such as the Global Environmental Fund and FAO (Food and Agriculture Organization of the United Nations). IPBES works with them, including during outreach and dissemination activities;

2) Broader audiences:

a) Scientific community: IPBES depends on the scientific community for the production of its reports and should therefore target this community to increase its engagement. International associations of scientists could be targeted as part of outreach activities;

b) Indigenous and local knowledge holders and experts: The IPBES commitment to use multiple knowledge systems makes both communities important target audiences;

c) Business and industry: it is anticipated that IPBES's reports will be considered by businesses and industries to help find sustainable ways of avoiding, minimizing, mitigating and offsetting impacts on ecosystems;

d) Practitioners or implementers: a multitude of organizations and individuals involved in the implementation of programs depending on or affecting biodiversity and ecosystem services working on the ground will be interested in learning about the products of IPBES, such as policy support tools, and how they can use them;

e) Community-based organizations: certain communities, including environmental non-governmental organizations. will be greatly affected by biodiversity loss and/or committed to its rehabilitation, and will therefore need to be aware of the findings of IPBES's assessments and policy support tools. The IPBES Secretariat could work with relevant networks to disseminate communications materials to these communities;

f) Intergovernmental and non-governmental organizations: these may be able to support IPBES's objectives by providing outreach to their constituencies, including policymakers or the private sector, and by using the networks connected to their respective National Focal Points;

g) Funding agencies that support national, regional and international activities and may play crucial roles in enabling the actions of other target audiences on the list;

h) The media: the IPBES Secretariat would not be in a position to reach all audiences directly and would therefore rely on good media relations to reach broader audiences;

i) Communities and the public at large.

All these categories of target audiences may act as both contributors to and end users of IPBES outputs. All of them may:

i. Contribute to the activities of the work programme through their experience, expertise, knowledge, data, information and capacity-building experience;

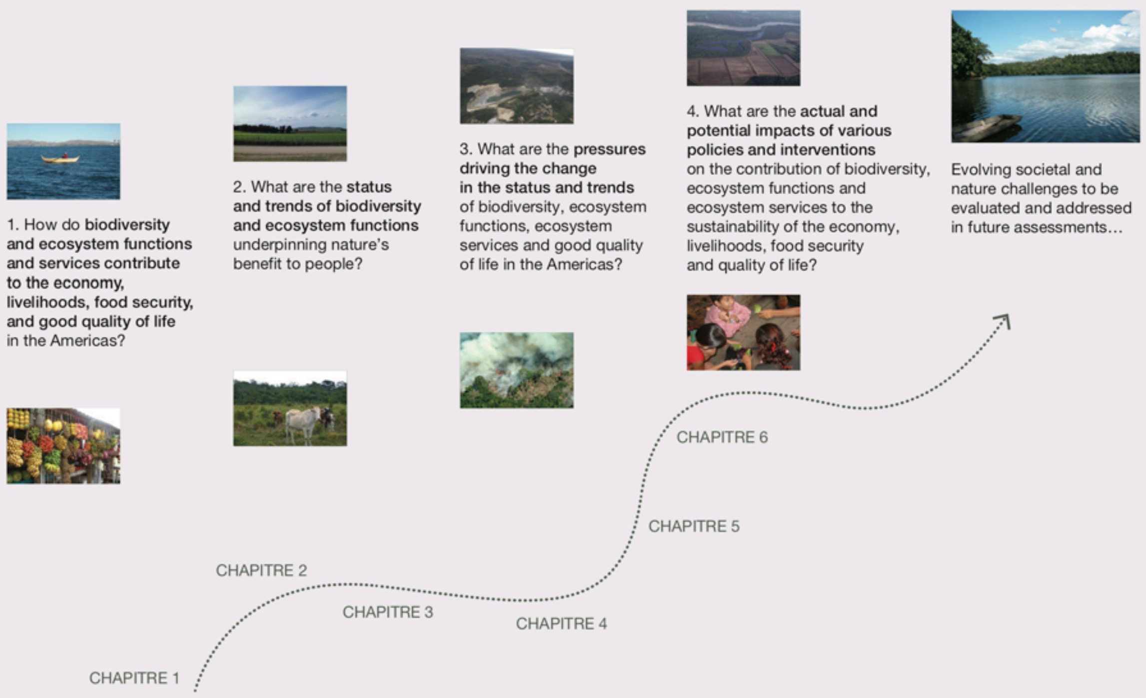
ii. Use or benefit from the outcomes of the work programme;

iii. Encourage and support the participation of scientists and knowledge holders in the work of the Platform.

1.4 Roadmap to core questions and chapters in this Regional Assessment

Chapter 1 sets the scene, and presents the policy-relevant questions identified for the region, subregions, units of analysis, and the IPBES conceptual framework used in the Americas Regional Assessment. The analysis in the remaining chapters is conducted to address those policy-relevant questions posed by governments and other decision makers (Figure 1.2) within the IPBES framework, which was designed to help address the science-policy interface on biodiversity and ecosystem services topics (Diaz et al., 2015).

Figure 1 2 Roadmap of policy-relevant questions addressed by all chapters of the Americas regional assessment. Source: own representation. Photo credits: Geraldo Arruda Junior, María Paula Barral, Margie Moss, Vanesa Rodríguez-Osuna and Nathan Vogt.



Chapter 2 is the primary place where the key following policy-relevant question is addressed*:* (1) How do biodiversity and ecosystem functions and services contribute to the economy, livelihoods, food security, and good quality of life in the regions, and their interlinkages?

It assesses the values of nature's contributions to people, the dependence or interrelationship of human well-being on biodiversity and NCP, information on the trends in human-wellbeing, and links those to trends in NCP. This chapter most explicitly draws on the diversity of knowledge systems, including Indigenous and local knowledge in addition to "western science". Also, in this Assessment, the concept of good quality of life is central to this Chapter, and continues as a thread through the subsequent chapters.

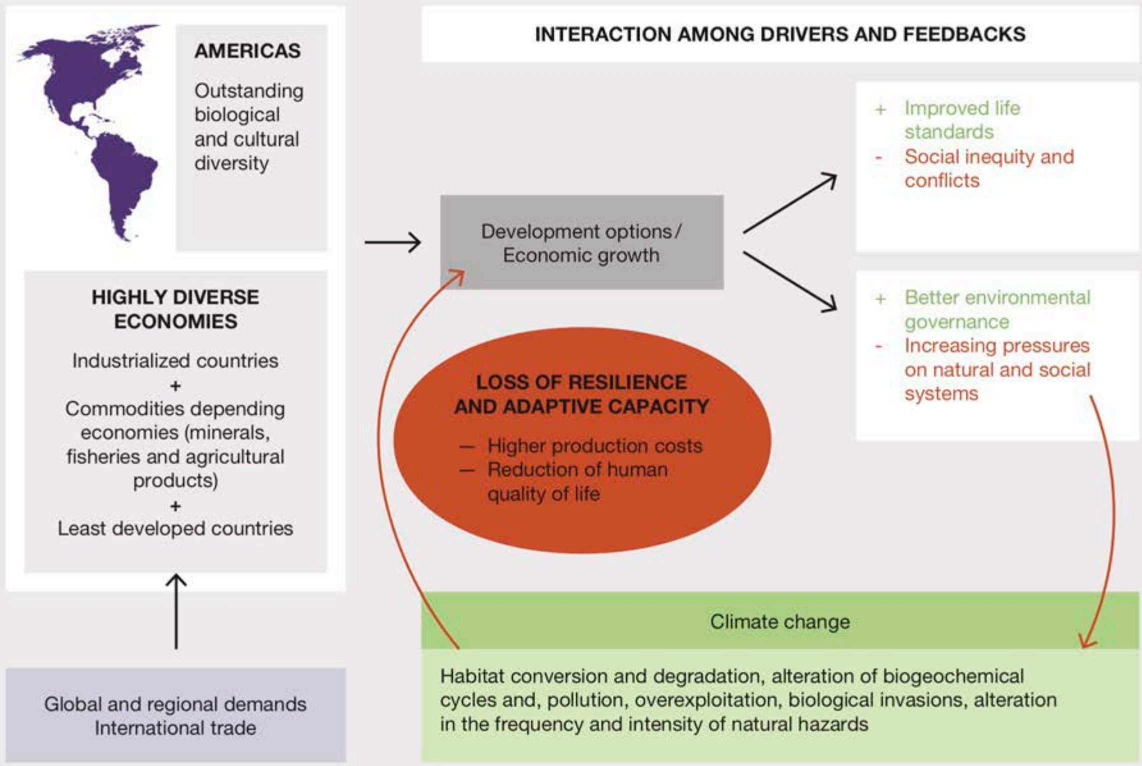
Chapter 3 focuses on the status and trends of biodiversity and ecosystem functions underpinning nature's benefit to people considering both structural and functional features of the biotic communities and their abiotic environments. It is the central place where the following policy-relevant question is addressed: (2) What are the status, and trends of biodiversity, ecosystem functions that ultimately affect their contribution to the economy, livelihoods and well-being in the region?

Chapter 3 assesses the amount of biodiversity found in the Americas, considering native and non-native biodiversity, how it is distributed across the Americas, the present state of ecosystems and biomes, recent changes in ecosystems and their biodiversity, the conservation status of species, and trends in levels of protection. It also provides an overview of the relative important of the units of analysis by subregion with regard to NCP. Additionally, the state of key ecosystem functions is assessed where information is available.

Chapter 4 focuses on drivers of changes in biodiversity and addresses the policy question: (3) What are the pressures driving the change in the status and trends of biodiversity, ecosystem functions, ecosystem services and good quality of life in the region?

This chapter presents information on status and trends of the factors that have potential to drive changes in biodiversity components, and consequently in the NCP. Chapter 4 reaches back to Chapter 3 for linkages of the drivers to biodiversity trends, and forwards to Chapters 5 and 6 for evaluation of alterative options for the intensity of the drivers. Where possible, it reaches toward finding evidence of possible indirect links between specific drivers and the trends in NCP described in Chapter 2.

Figure 1 3 Direct and indirect drivers of NCP in the Americas and their interdependencies. Source: own representation.



Chapter 5 provides a synthesis of the information contained primarily in chapters 2-4 and makes use of scenarios and modelling developed for the Americas Region. In this synthesis, the Chapter examines how the core questions 1-3 interact to affect human well-being (5.4). In particular, it examines the future trends of biodiversity and drivers and what those trends might mean in terms of the archetype scenarios of “business as usual” and “great transitions” (5.4, 5.5.1). Additionally, the Chapter examines the role and significance of telecoupling (5.6.1) and presents key findings on both telecoupling and data gaps (5.8), especially with respect to time series data on status of biodiversity and drivers. To the extent possible, the chapter explores changes in the trajectories of multiple drivers and the role played by synergies, trade-offs and adaptive behaviour.

Chapter 6 takes note of how the linkages and scenarios in earlier chapters may be facilitated or impeded by various policies options. It is where key question 4 is addressed: (4) What are the actual and potential impacts of various policies and interventions on the contribution of biodiversity, ecosystem functions and ecosystem services to the sustainability of the economy, livelihoods, food security and good quality of life in the region?

This chapter provides information to identify policies that may respond effectively to trends in biodiversity, NCP or human well-being. All chapters strive to present information in ways that are relevant to policy-making but not prescriptive regarding choices among policies and options for decision makers at the regional and subregional levels in response to the scenario set out in previous chapter. Chapter 6 also explores the policy framework available and their track record in the Americas. To the extent possible many of the social, economic, cultural and governance factors that affect their performance are considered.

1.4.1 What gaps in knowledge need to be addressed to better understand and assess drivers, impacts and responses of biodiversity, ecosystem functions and services at the regional level?

Much biodiversity remains to be scientifically under sampled for all types of ecosystems in the Americas, particularly in South America and in the deep oceans. The potential areas with gaps in knowledge in this Regional Assessment include:

i. the contributions of NCP to quality of life, considering the mismatch of social and quality of life (well-being) data produced at the political scale and ecological data produced at a biome scale;
ii. the assessment of non-material NCP that contribute to quality of life,
iii. the linkages from indirect to direct drivers and from the drivers to specific changes in biodiversity and NCP,
iv. the factors that affect the ability to generalize and scale up or down the results of individual studies, and
v. the evaluation of the impacts of short-term and long-term policy and programmes.
vi. Investments in generating new knowledge on these matters, which are discussed across chapters, may better elucidate how human quality of life is highly dependent on a healthy natural environment as well as how threats to natural environments affect quality of life in the short, median and long-term.

1.4.2 Relationship of the key questions to the implementation of the Strategic Plan for Biodiversity and its Aichi biodiversity targets and to the Sustainable Development Goals

The Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services Assessments consider the synergies and trade-offs associated with meeting multiple goals and the interactions among the social (including cultural), economic and environmental dimensions of sustainable development. This Regional Assessment is highly relevant in the context of the CBD Strategic Plan for Biodiversity 2011–2020 and Aichi biodiversity targets, as well as national biodiversity strategies and action plans, and the UN Sustainable Development Goals for 2030. The CBD strategic plan and targets are products of this convention’s negotiations while the SDG resulted from the entire UN level negotiations agreed upon 193 countries.

In this Regional Assessment, the time frame of analyses covers the current status, trends up to 2020 (going back as far as 50 years) and plausible future projections, with a focus on various periods between 2020 and 2050 that cover key target dates related to the Strategic Plan for Biodiversity 2011–2020 and the SDG. The analyses include an evaluation of the likelihood of achieving the targets and goals (Chapters 2-6) if present trends continue, and identify the types of changes in trends of biodiversity, and the drivers of those trends that would increase the likelihood of achieving targets and goals that at present may appear elusive.

The degree of government’s commitment to conservation and sustainable use of biodiversity are captured partly in the endorsement of many global agreements and conventions about biodiversity and its uses, presented in Table 1.2. For most countries, global commitments are often uncoupled from national policies (6.3).

Table 1.2. Countries participating in international environmental commitments by subregion

Convention name		North America-2*	Mesoamerica-8	South America-12	Caribbean-13*
Convention on Biological Diversity (CBD)		1	8	12	13
United Nations Convention on the Law of the Sea (UNCLOS)		1	8	9	13
Paris Accord (United Nations Framework Convention on Climate Change)		2	7	10	13
CITES (Convention on International Trade in Endangered Species of Wild Fauna and Flora)		2	8	12	11
United Nations Conventions to Combat Desertification (UNCCD)		2	8	12	13
Convention on the Conservation of Migratory Species of Wild Animals		0	3	12	2
Ramsar Convention		2	8	11	8
Percentage of Area Protected	Terrestrial	14.40	28.20	25	14.60
	Marine	6.90	2.10	3.90	1.20

*Greenland and the 13 Caribbean Island Protectorates still have aspects of foreign policy such as becoming Parties to international agreements and conventions, influenced by other soverign States, and are not included in this table. Source: Own representation and percentage of area protected from Juffe-Bignoli *et al.* (2014).

In the Americas, all countries, with the exception of the USA, are signatory to the CBD. Results from the 24 countries of the Latin America and Caribbean regions have reported mixed levels of progress towards the biodiversity 2020 Aichi targets. Most progress has been reported in targets 11 and 17 (Protected areas and the adoption and implementation of policy instruments). There is evidence of advanced progress in target 1 (People being aware of the value of biodiversity and the steps to conserve and use it sustainably); target 16 (Nagoya Protocol) and target 19 (Improved biodiversity information sharing). The targets reporting less progress were targets 6 (Anthropogenic pressures/direct drivers of change minimized) and 10 (Management of fish and aquatic invertebrate stocks) (Chapter 6).

Even at these early stages of the sustainable development agenda, SDG are already providing essential policy entry points to address a broad array of drivers that affect biodiversity and ecosystem services (Chapter 6).

Given the negative impacts of policy choices and trade-offs on some aspects of biodiversity and NCP and quality of life, few of the Aichi biodiversity targets will be met by 2020 for most countries in the Americas. In a longer term perspective, few SDG or targets set under the Paris Agreement are likely to be met under current business as usual scenarios (Chapters 2-3).

1.5 The conceptual approach for this Assessment

For an assessment to address the many types of issues encompassed in the IPBES core questions in section 1.4 and be of use to the broad range of target audiences described in 1.3.6, it must have as well structured foundation. Integrative but explicit conceptual frameworks are particularly useful tools in fields requiring interdisciplinary collaboration, where the frameworks are used to make sense of complexity by clarifying and focusing thinking about relationships, supporting communication across disciplines and knowledge systems and between knowledge and policy. This foundation is provided by the IPBES Conceptual Framework.

1.5.1 The analytical Intergovernmental Platform on Biodiversity and Ecosystem Services Conceptual Framework

The Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services has developed a conceptual framework (CF, Figure 1.4) as a concise summary of the relationships between people and nature in words and pictures. The framework provides a common terminology and structure for the components that are the focus of a system analysis, and proposes assumptions about key relationships in the system.

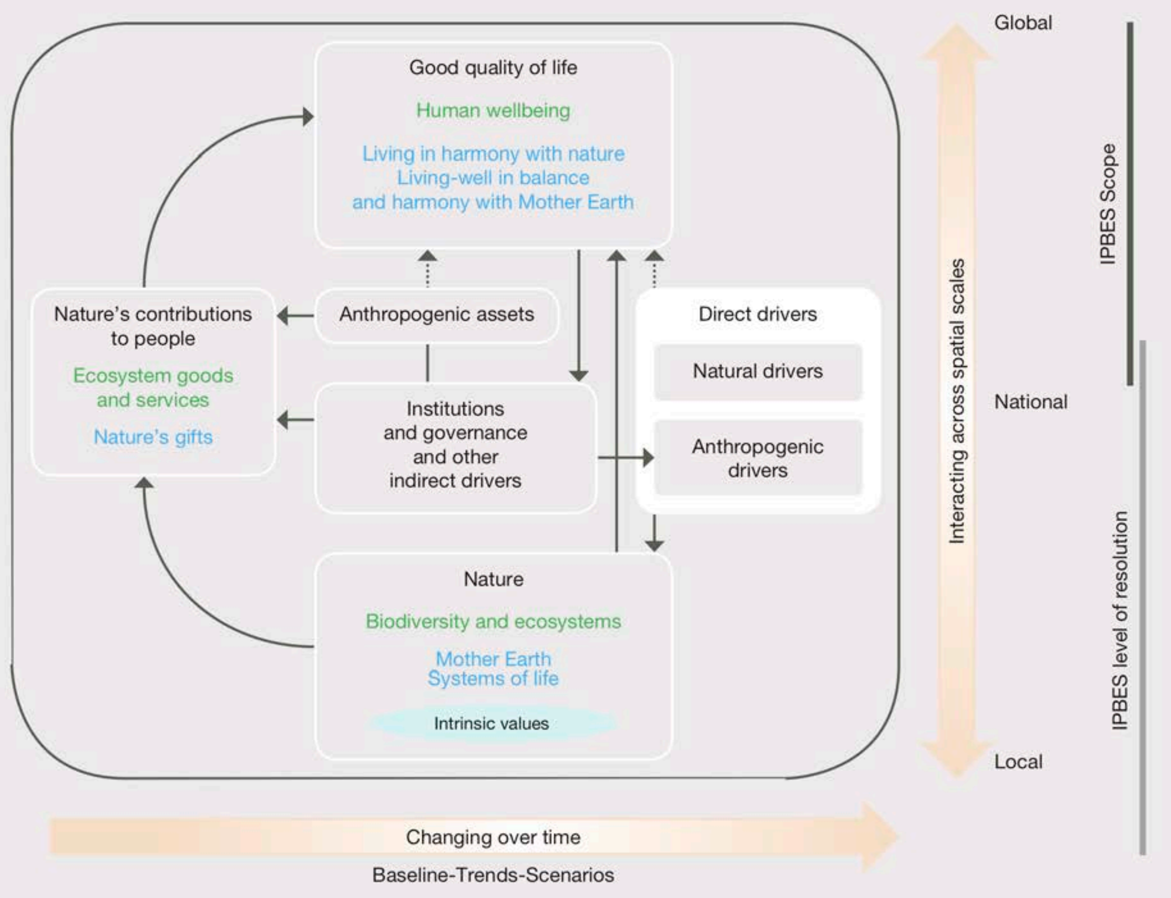
The main elements of the IPBES Conceptual Framework

- Nature here refers to the natural world, with an emphasis on biodiversity and ecosystems. Nature gains values based on the provision of various benefits to people, but within IPBES Assessments, nature is also recognized as having intrinsic value, independent of human experience.
- Anthropogenic assets refer to knowledge, technology, financial assets, built infrastructure, etc.
- Nature's contributions to people is, for IPBES, an inclusive category across knowledge systems. It is defined as "all the benefits (and when they occur, losses or detriments) that humanity obtains from nature" (Pascual et al., 2017; sections 1.3.1-1.3.2)
- Institutions and governance systems and at least some other indirect drivers are fundamentally linked to the direct anthropogenic drivers that affect nature. They include systems of access to land, legislative arrangements, international regimes such as agreements for the protection of endangered species, and economic policies.
- Direct drivers, both natural and anthropogenic, affect nature directly. The direct anthropogenic drivers are those that flow from human institutions and governance systems and other indirect drivers. They include positive and negative effects, e.g. habitat conversion (e.g. degradation or restoration of land and aquatic habitats), climate change, and species introductions. Direct natural drivers (e.g. volcanic eruptions) can directly affect nature, anthropogenic assets, and quality of life, but their impacts are not the main focus of IPBES.
- Indirect drivers, are the ways in which societies organize themselves, and the resulting influences on other components. They are the underlying causes of environmental change that are exogenous to the ecosystem in question. Because of their central role, influencing all aspects of human relationships with nature, these are key levers for decision-making.
- Good quality of life is the achievement of a fulfilled human life. It is a highly value-based and context-dependent element comprising multiple factors such as access to food, water, health, education, security, cultural identity, material prosperity, spiritual satisfaction, and freedom of choice. A society's achievement of good quality of life and the vision of what this entails directly influences institutions and governance systems and other indirect drivers and, through them, all other elements. Good quality of life, also indirectly shapes, via institutions, the ways in which individuals and groups relate to nature. Likewise, the institutions and governance systems can be used by people to influence a society's value system and perception of what constitutes good quality of life". IPBES does not address this aspect of the conceptual framework in the Assessments, but actions governments and societies may choose to take based on the findings of the IPBES Assessments often require addressing this pathway wisely.

Within these broad and cross-cultural categories, the Assessment identifies more specific subcategories, associated with knowledge systems and disciplines in the Americas. For example, different worldviews may have large gaps between the ways in which ecosystem goods and services ("green" category) and contributions of nature ("blue" category) in Figure 1.4 are conceptualized, valued and used accordingly. However, both categories are concerned with the things that societies obtain from the natural world, which are collectively represented by the inclusive category nature's contributions to people ("bold and black" category). For consistency across Assessments, and to follow the spirit of the conceptual framework, the

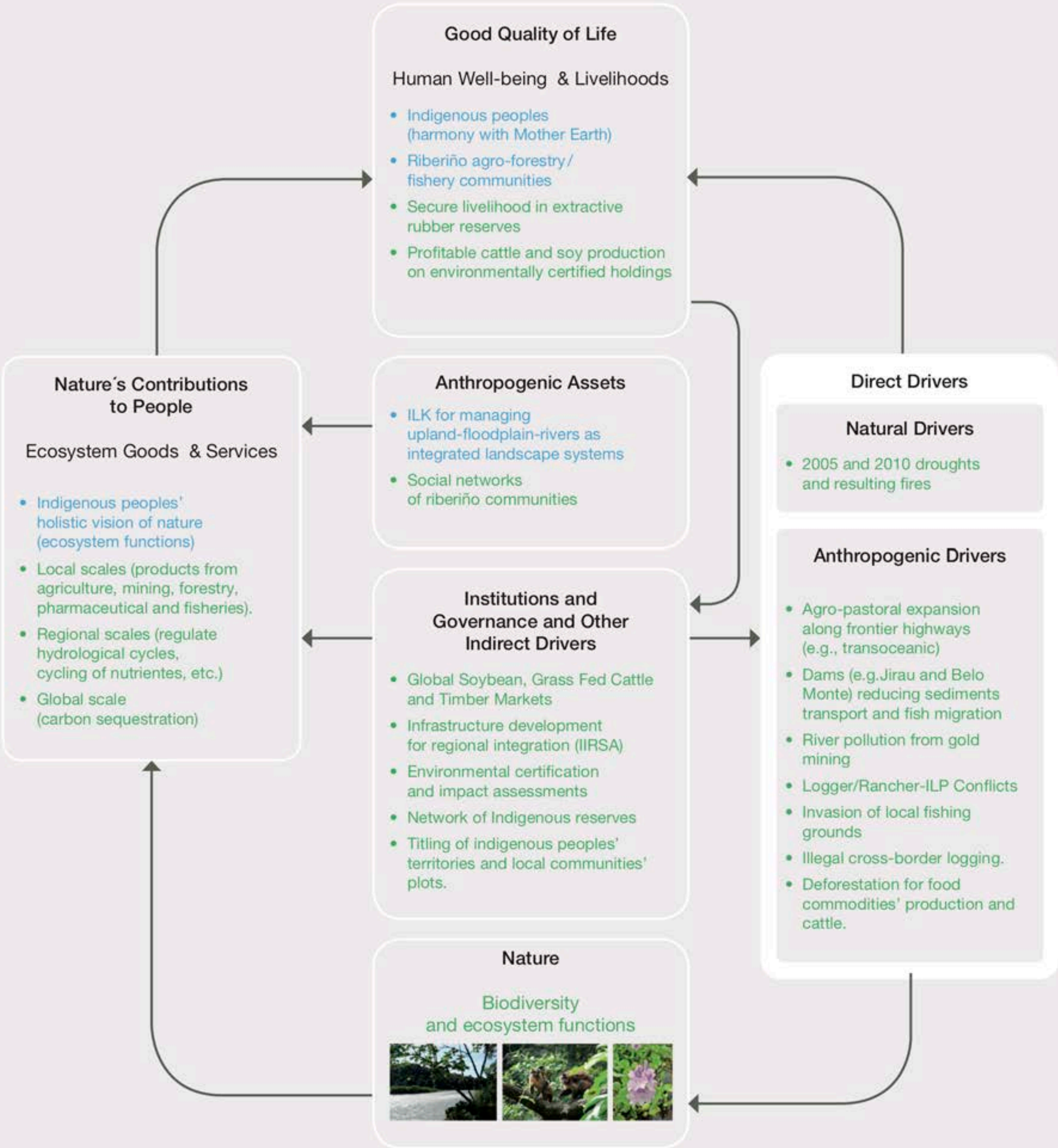
Assessments will use the inclusive "bold and black" categories as the starting point, and then refer back to them in the conclusions, although more specific categories, strongly dependent on discipline, knowledge system and purpose are likely to be used in the analytical work during the Assessment. The use of this conceptual framework is presented in an example in the Amazon region in Figure 1.5.

Figure 1 4 **The analytical Conceptual Framework from IPBES. Source: modified from Diaz *et al.* (2015).**



In the main panel, delimited in grey, boxes and arrows denote the elements of nature and society that are the main focus of the Platform. In each of the boxes, the headlines in black are inclusive categories that should be intelligible and relevant to all stakeholders involved in IPBES and embrace the categories of western science (in green) and equivalent or similar categories according to other knowledge systems (in blue). The blue and green categories mentioned here are illustrative, not exhaustive, and are further explained in the main text. Solid arrows in the main panel denote influence between elements; the dotted arrows denote links that are acknowledged as important, but are not the main focus of IPBES. The thick coloured arrows below and to the right of the central panel indicate that the interactions between the elements change over time (horizontal bottom arrow) and occur at various scales in space (vertical arrow). The vertical lines to the right of the time arrow indicate the geographical scale (scope), build on properties and relationships acting at finer (national and subnational) scales (resolution). The resolution line does not extend all the way up to the global level because, for the types of relationships explored by IPBES the spatially heterogeneous nature of biodiversity is important, so IPBES Assessments will be most useful if they retain finer resolution.

Figure 1 5 NCP in the Amazon: Applying the IPBES Conceptual Framework. Source: own representation.



1.5.2 How this Regional Assessment deals with different knowledge systems

Scientific knowledge, indigenous knowledge, and local knowledge systems all play a central role in IPBES Assessments. In IPBES, indigenous and local knowledge (ILK) systems are defined as dynamic bodies of integrated, often holistic, social-ecological knowledge, practices and beliefs about the relationship of living beings, including humans, with one another and with their environment. Indigenous and local knowledge is highly diverse, produced in a collective manner and reproduced at the interface between the diversity of ecosystems and human cultural systems. It is continuously evolving through the interaction of experiences and different types of knowledge (written, oral, tacit, practical, and scientific) among indigenous peoples and local communities.

Governance, institutions and policies vary in the extent and ways that they take into account indigenous and local knowledge and practices (Pascual et al., 2014; Martin et al., 2016; Vogt et al., 2016). Indigenous and local knowledge can take a particularly prominent role when addressing "values" and valuation in

Assessments. Valuation tools that use multiple knowledge systems to fully capture the multiplicity of culturally different worldviews, visions and approaches to achieving a good quality of life are needed and often not available (Tengö et al., 2014). To this end, IPBES has developed a preliminary guide on the diverse values of nature and its contributions to people. This guide complements guidance IPBES has developed for the integration of ILK into its Assessments that respects not only the diversity and value of ILK, but also the rights of indigenous and local communities to share in the benefits of knowledge gained from the Assessmentss (Pascual et al., 2014; Berbes-Blazquez, 2016). IPBES integrates ILK into its Assessmentss through the appointment of experts to conduct and review Assessmentss represent, who can or have expertise, in ILK.

1.5.3 How this Regional Assessment deals with "value"

Understanding values, how they are conceptualized and formed and how they change across contexts and scales, is critical to inform decision making and policy design at local, national and global levels (IPBES, 2015b). The ways in which nature and its contributions to people for a good quality of life are perceived and valued may be starkly different and even conflicting (IPBES, 2015b; Pascual et al., 2017). Multiple values can be associated with multiple cultural and institutional contexts and may be often difficult to compare by the same measure. Therefore, IPBES recognizes that the word 'value' can refer to a given worldview or cultural context, a preference someone has for a particular state of the world, the importance of something for itself or for others (IPBES, 2015b; Pascual et al., 2017).

At present, governance, institutions and policies are challenged to take adequately into account the diverse conceptualization of multiple values of nature and its contributions to people embodied in the IPBES conceptual framework (Pascual et al., 2017). Any single valuation methodology applied to NCP cannot avoid reflecting the values attached to the specific uses to be made by the NCP, and those uses vary widely among cultures, societies and economic strata. Therefore, if valuation is intended to encompass diverse perspectives, a multiplicity of valuation methodologies will be needed, as well as methods for combining the results in ways that do not selectively favour one worldview over other. Such methodologies and strategies for combining results are not yet fully developed. Nevertheless, assessments striving to move in that direction can be a significant resource for a range of decision makers, including governments, civil society organizations, indigenous peoples and local communities. Therefore, IPBES Assessmentss will be based on the recognition of culturally different worldviews, visions and approaches to achieving a good quality of life in the context of the conceptual framework (section 1.5.1 presenting results of using multiple approaches to valuation, and interpreting the results in inclusive contexts).

1.5.4 How can models and scenarios serve as tools for decision-making?

Scenarios are plausible, challenging, and relevant stories about how the future might unfold, while a scenario archetype is a group of futures which are deemed 'similar' according to the purpose of a specific analysis (Boschetti et al., 2016). The different scenarios in a set can reflect different plausible future trajectories of indirect and direct drivers of nature and NCP; responses to potential policy and management interventions; or the results of a combination of these (IPBES, 2016b). Models refer to qualitative, or when possible quantitative, descriptions of the links between any two elements of the framework that provide the means to relate changes in one element to estimates, or projections, of changes in the other.

Scenarios and models can provide an effective means of gaining insight into relationships among nature, nature's contributions to people, and good quality of life according to different worldviews. For example, we can analyze different scenarios of access to land impact well-being of indigenous communities (given the dependence of these actors on certain components of biodiversity such as food and medicinal plants, see chapter 2), show how those same scenarios affect differently other actors such as agricultural producers, or inform discussions of both perspectives.

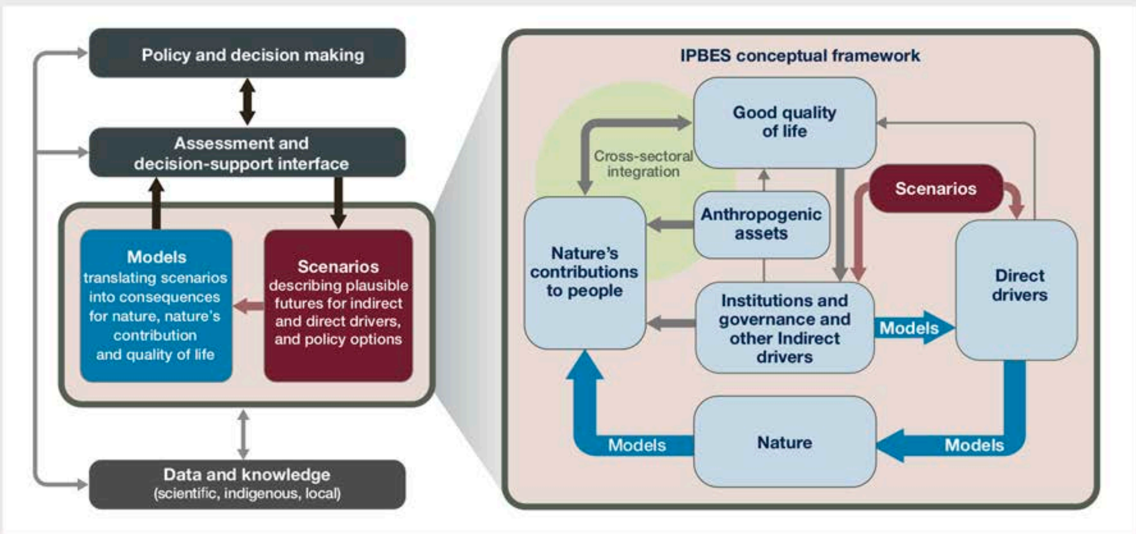
One of the key objectives in using scenarios and models is to move away from a reactive mode of decision-making, in which society responds to the degradation of biodiversity and nature's benefits to people in an uncoordinated, piecemeal fashion, to a proactive mode, in which society anticipates change and takes actions that avoid, reduce or mitigate adverse impacts, capitalizes on important opportunities, and ensure adaptation and mitigation strategies are integrative and holistic (Carpenter et al., 2006). Scenarios and models used in IPBES are typically explicitly or implicitly built on four main components:

- Scenarios of socioeconomic development (e.g. population growth, economic growth, per capita food consumption, greenhouse gas emissions) and policy options (e.g. reducing carbon emissions from deforestation and forest degradation, subsidies for bioenergy);
- Projections of changes in direct drivers of biodiversity and ecosystem function (e.g. land use change, fishing pressure, climate change, invasive alien species, nitrogen deposition);
- Projections of the impacts of drivers on biodiversity (e.g. species extinctions, changes in species abundance and shifts in ranges of species, species groups or biomes);
- Projections of the impacts of drivers and changes in biodiversity on NCP (e.g. ecosystem productivity, control of water flow and quality, ecosystem carbon storage, cultural values).

These elements generally correspond to the structure of the IPBES conceptual framework, and Figure 1.6 below illustrates how scenarios and models are typically coupled to provide projections of future trajectories of biodiversity, NCP and human well-being. Elements can range from highly quantitative (e.g. econometric models of socioeconomic development) to qualitative (e.g. prospective scenarios of development based on expert-stakeholder dialogues (Coreau et al., 2009).

The Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services aims to match its scenarios carefully to the needs of particular policy or decision contexts, paying particular attention to (i) the choice of drivers or policy options that determine the appropriate types of scenarios (e.g. exploratory, target-seeking or policy screening); (ii) the impacts on nature and nature's benefits that are of interest and that determine the types of models of impacts that should be mobilized; (iii) the diverse values that need to be addressed and that determine the appropriate methods for assessing those impacts; and (iv) the type of policy or decision-making processes that are being supported and that determine the suitability of different assessment or decision-support tools (e.g. multi-criteria analysis and management strategy evaluation).

Figure 1 6 The role of scenarios and models in IPBES assessments. Source: modified from IPBES (2016).



1.5.5 Impact of policies on nature's contribution to people

Policies can affect ecosystem structure, functions and ecosystem services (NCP) by altering how governments, institutions, and individuals interact with nature. Policies are designed to address particular challenges such as the loss of biodiversity and ecosystem services using different types of tools or instruments.

Some policy tools provide incentives for behaviors that are consistent with restoring or maintaining ecosystems or disincentives to behaviors that can lead to harmful impacts on ecosystem structure or function or availability of NCP (e.g. fines and taxes). Policies can indirectly affect the value decision-makers or citizens give to ecosystems by providing incentives, disincentives or enabling conditions directed at the actions of civil society, the corporate community, and government institutions. For example, policy instruments such as legally protected lands can affect positively the value of these areas for their supply with drinking water and associated NCP by protecting its quality and quantity, Reciprocally, if people place a high value on experiencing natural areas (also an NCP), they can provide incentives for decision-makers to support policies that protect such areas (WB & WWF, 2003). In Venezuela, the economic value of the reduced sedimentation from a national protected area system is estimated at approximately $3.5 million annually (in terms of reduced farmer income) (Pabon-Zamora et al., 2008). However, if not designed and implemented carefully, such benefits may come at the cost of displacement of local community uses of protected areas, such as when marine protected areas attract significant ecotourism revenues, but displace community-based fisher families with few alternative options for livelihoods (FAO, 2015).

On the other hand, policies may result in incentives to use biodiversity and ecosystem services (nature and NCP) irresponsibly. For example in the energy sector, domestic subsidies of fuel prices promote overutilization of these resources, increases greenhouse gas emissions, and a negative contribution to climate change (IEA, 2015) accelerating climate change impacts on biodiversity and people (Bruckner et al., 2014). Alternative policies such as decarbonizing electric generation, applying carbon standards to power plants or eliminating subsidies for producing or consuming fossil fuels may have different consequences, including reducing air pollution (Schwanitz et al., 2014) and their associated benefits to human health (Buonocore et al., 2015; Driscoll et al., 2015); improving energy efficiency (IEA, 2015) and developing renewable energy sources (Bruckner et al., 2014). However, such alternatives must be considered fully, as hydroelectric power may require substantial modifications to natural watersheds, and mining the raw materials needed for solar panels can have a large environmental footprint (Bruckner et al., 2014; Nugent & Sovacool, 2014). These complexities in developing responsible policies for conservation and sustainable use of nature and NCP highlight the importance of the efforts of the IPBES Regional Assessments to consider the multiple knowledge systems and the values of diverse worldviews, and to use scenarios and models effectively.

Regional differences also influence in the way some policies affect value given to ecosystems, for example to protected areas and their relation to ecotourism. Policies addressing ecotourism could emphasise the substantial economic benefits from recreational use associated with ecotourism in conserved areas or give more weight to protective approaches to biodiversity conservation and restrict ecotourism stringently.

Similarly, policies and values for food production systems can either promote genetically modified crops grown with highly industrialized production systems, or favour production systems using traditional varieties of plants involving rich local and indigenous knowledge applied to the cultivation of such plants under particular environmental settings (Jacobsen et al., 2013; Bazile et al., 2016; CIP, 2017).

Current dialogue on NCP emphasizes the importance of their relationships with livelihoods and human well-being (Raudsepp-Hearne et al., 2010; Haines-Young & Potschin, 2012), interactions among multiple services (Kremen, 2005; Bennett et al., 2009; Rodríguez Osuna et al., 2018), how bundles of NCP can help us understand co-benefits and trade-offs (Raudsepp-Hearne et al., 2010), and that some contributions accrue to private beneficiaries in contrasted with broader public goods (Garbach et al., 2014, 2016). Policies and programmes that are able to adopt bundling approaches to NCP, where multiple benefits and trade-offs are measured and assured (e.g. water and food security, climate change adaptation as well as social and cultural benefits) provide opportunities towards the achievement of sustainable development and biodiversity goals.

1.6 Nature and economies of the Americas

1.6.1 Biophysical aspects

The Americas encompass a large diversity of ecosystems, including wide extensions of deserts, grasslands, savannas and forests, in different climatic conditions (polar, temperate, mediterranean, arid, subtropical, tropical) and topographic situations (plains, plateau, mountains). The combination of all those settings along the Neotropic and Neartic biogeographical realms covers all the 14 terrestrial biomes defined by Olson et al., (2001), as well as all the freshwater and marine biomes defined in the Marine Ecosystems of the World and Global Open Ocean Deep Sea classifications (Spalding et al., 2007; Rice et al., 2011). The region includes also 55 of the 195 terrestrial and freshwater ecoregions considered globally as having exceptional biodiversity, i.e. with highly distinctive or irreplaceable species composition (Olson & Dinerstein, 2002), including the largest rainforest and largest river in the world situated in the Amazonian region. Similarly, the Caribbean is considered a hotspot for marine biodiversity, as are reefs and bays of Mesoamerica (WOA, 2016).

The Intergovernmental Platform on Biodiversity and Ecosystem Services unit of analysis and subregions of the Americas

The subdivision of the Earth's surface into units for the purposes of analysis is notoriously controversial and there is no single agreed-upon system that IPBES can adopt as its standard. The Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services has consulted widely to arrive at the classification below. This system serves as a framework for comparisons within and among assessments and represents a pragmatic solution, which may be adapted and evolve as the work of IPBES develops. These units are called "IPBES terrestrial and aquatic units of analysis" (Figure 1.7), rather than alternatives such as "biomes" or "ecoregions", both because they do not map exactly onto such ecological classifications, and among different disciplines there is rarely consensus on the geographic boundaries when applying such classification systems. These units of analysis serve the purposes of IPBES, and are not intended to be prescriptive for other purposes; nor are the labels of individual units to be taken as synonymous with "biomes" from any single classification system. Note also that the word "aquatic" is here used to include both marine and freshwater systems.

Ecological units of analysis are represented in different socio-economic and governance contexts with different administrative boundaries. For this reason, IPBES has also decided to use a classification of the Americas in four subregions considering their focus on science-policy interface (Figure 1.8).

Figure 1 7 **Units of Analysis of the Americas assessment. Source: own representation based on Olson (2001), WWF (2004 and 2012) and Marine Regions (2016).**

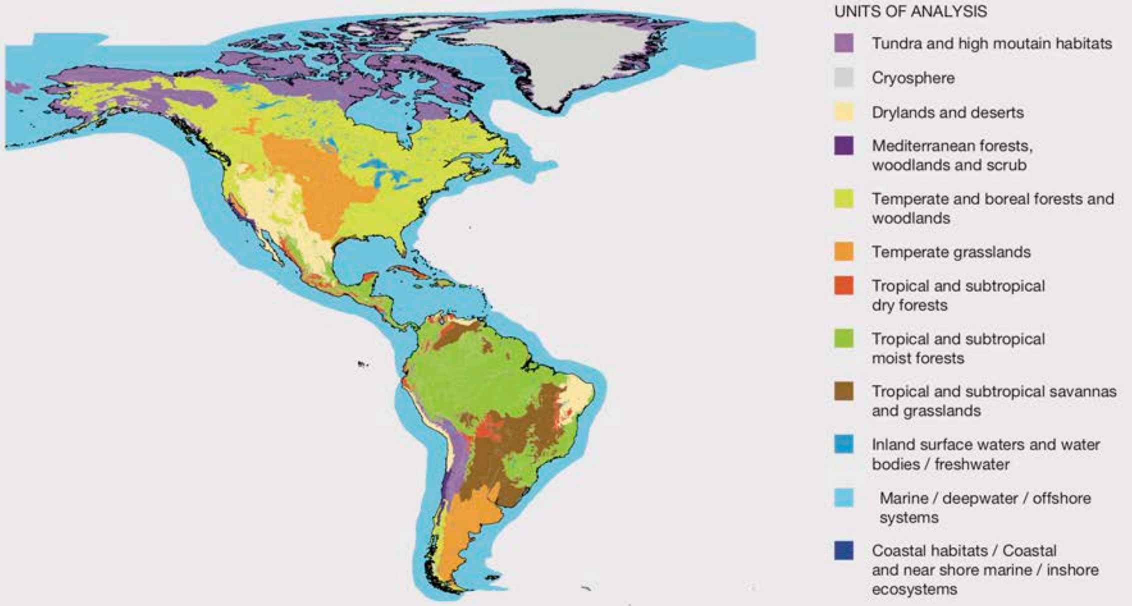
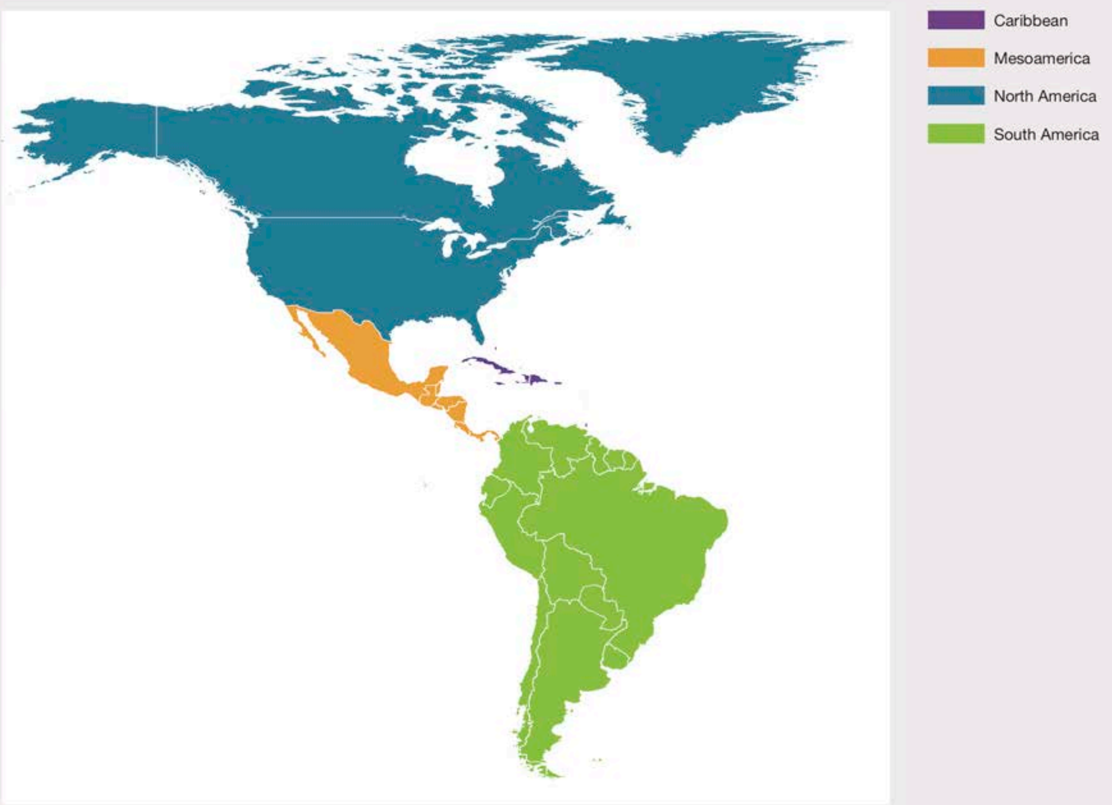


Figure 1 8 **Subregions of the Americas assessment. Source: own representation based on the map downloaded from http://www.naturalearthdata.com/.**



North America

North America is the largest subregion of the Americas, at just over 23.5 million km^2. At the time of European settlement starting in the 1500's, all major temperate and polar units of analysis were extensive and intact. The eastern third of North America was dominated by temperate, primarily deciduous, forests covering all the coastal lowlands, the Appalachian mountains (only of few of which extend above the treeline) and the eastern portion of the Mississippi River basin. Across the northern portion of treed lands, boreal forests constituted a band often nearly 1,000 km wide, extending from the Atlantic to the Rocky Mountains and Alaska. The central portion of North America comprised the Great Plains and related grasslands, covering nearly 1.3 million km^2 of unbroken grassland. The western Rocky Mountains and Pacific Coastal Range along the Pacific seacoast, both extending from Mexico to Alaska, together covered over 1.5 million km^2. In the USA southwest more than 0.75 million km^2 were drylands and desert, whereas the world's largest expanse of tundra was found across the entire northerly continental land mass and Arctic Archipelago of Canada and Greenland (including the ice sheet and glaciers), at nearly 3.5 million km^2.

Several major river systems, and many smaller ones, drain North America, emptying into the three bordering oceans. The largest is the Mississippi-Missouri drainage flowing southward through the center of North America to the Gulf of Mexico. With a drainage area of over 3 million km^2 it is the fourth largest drainage basin in the world. Also flowing southward but into the Gulf of California and draining much of the desert southwest is the Colorado River basin. The Great Lakes, the largest freshwater lacustrine system in the world, are part of the easterly flowing St Lawrence River drainage, emptying through the Gulf of St Lawrence into the Atlantic. The major rivers flowing northward into the Arctic Ocean are the Mackenzie and the Yukon, whereas the largest river drainage emptying directly into Pacific Ocean is the Columbia. Aside from the Mackenzie and Yukon, all these river systems have been extensively altered for navigation, hydropower generation, flood control, municipal water supply, and irrigation.

With the expansion of settlement by non-indigenous immigrants and their descendants, most of these biomes were extensively altered through land transformation and development of urban areas and linking infrastructure. With the changes in landforms, many iconic species, such as the American Bison and Pacific salmon have declined or, in the case of the once abundant Passenger Pigeon, become extinct. The Indigenous Peoples inhabiting these biomes were also decimated by conquest, disease, and intentional displacement from traditional lands, although the precise numbers are contested among experts, and their traditional livelihoods, closely attuned to nature and sustainable use of NCP, typically rendered impossible to pursue.

Mesoamerica

The Mesoamerican subregion is considered a priority ecoregion due to the high concentration of small-ranged vertebrates (Jenkins et al., 2013) and a biodiversity hotspot due to the high concentration of endemics species and large loss of habitat (Myers et al., 2000). This region connects species movement among south and north land masses resulting in high species diversity (DeClerck et al., 2010). Its particular long and narrow shaped area is divided by a mountain range creating diverse environmental conditions (Olson et al., 2001; DeClerck et al., 2010) with montane biomes extending along the entire south-north axis of Mesoamerica. Mangroves and coral reefs occur in patches along both Atlantic and Pacific coasts, although more extensively on the Atlantic. Reflecting the narrow width and central mountains of Mesoamerica, rivers are generally a most a few hundred km (Grijalva river), aside from the larger Rio Grande drainage on the northern boundary of the subregion (Lehner et al., 2006).

Ten per cent of the territory is under some form of protection (WDPA, 2017) where the 1) mediterranean forests, woodlands, and scrub, 2) tropical and subtropical dry broadleaf forests and tropical and 3) subtropical moist broadleaf forests are the least protected biomes.

The Mesoamerican subregion holds a very high level of endemism of 44.4%. Of these, over 40% are threatened. In total, 84.7% of all the subregion's threatened species are endemic. Particularly well-known subregional endemics include the Old Man Cactus (Ce*phalocereus senilis*) and the Axolotl (*Ambystoma mexicanum*).

Caribbean

The Caribbean Region comprises twenty-eight island nations which themselves are composed of over seven thousand islands and cays. As Small Island Developing States, these predominantly coastal areas are under risks from extreme geophysical events by virtue of their geographic locations within the tropics. They are susceptible to the hazards of hurricanes, earthquakes, volcanic eruptions and tsunamis (Granger, 1997). The islands are characterized into five (geophysical) categories: volcanic islands of recent formation; old complex volcanic islands; volcanic islands with lagoons and barrier reefs; atolls and raised atolls; and successive sedimentary deposit islands.

The steep topography seen on these islands supports a variety forest types in small areas (Lugo et al., 1981). These forests range from mangrove forests dominated by 1-4 mangrove species, to tropical rain forests comprising two thousand species of flowering plants (Beard et al., 1944). The Dry Forests in Puerto Rico, USA Virgin Islands and The Bahamas present a diverse and unique biome for the Caribbean Islands (Franklin et al., 2015). The Guanica forest in Puerto Rico comprises approximately four thousand hectares of dry forest (Lugo et al., 1995). As most of these dry forests are coastal, they are under increased risk of damage from hurricanes, storm surge and sea level rise.

The coral reef ecosystems that surround most of the islands of the Caribbean support the major sectors of tourism and fishing. However, these reef areas are under significant threat from overfishing and direct results of human activities causing excess nutrients and sediments via pollution, deforestation, reef mining and dredging (Hughes, 1994; Perry et al., 2013). The architectural complexity has declined over the past forty years (Alvarez-Filip et al., 2009).

South America

South America is the second largest subregion of the Americas, comprised of 12 States, covering 17.7 million km^2. South America exhibits a diverse pattern of weather and climate due to its considerable north-south extension and prominent topography, including tropical, subtropical and extratropical features. The large scale phenomena like the El Niño Southern Oscillation, contribute to the high variability of the South American climate (i.e. interannual and interdecadal changes), and the sea surface temperature north-south gradient has a profound impact on the climate and weather of eastern South America (Garreaud et al., 2009).

South America is characterized by the presence of the Andes, the longest continental mountain range in the world (Campetella & Vera, 2002). The Andes cover more than 2,500,000 km^2 hosting a population of about 85 million (45% of total continental population), with the northern Andes as one of the most densely populated mountain regions in the world. At least a further 20 million people are also dependent on mountain resources and ecosystem services in the large cities along the Pacific coast of South America. The Andes is highly diverse in terms of landscape, biodiversity including agro-biodiversity, languages, peoples and cultures (FAO, 2012a).

Another particularity of the region is the extensive watershed of big rivers, like Amazon, Orinoco, Paraná, among de various long rivers of South America (Nilsson et al., 2005). The largest is the Amazon Basin, containing forests that not only sustain the greatest biological diversity (Amazon is home to one out of every five mammal, fish, bird and tree species in the world); but the homes to indigenous peoples. At regional and global scales, tropical forests also have a major influence on carbon storage and climate, so they are also vital for regional climates (Laurence, 1999). The trees of the Amazon contain 90–140 billion tons of carbon, equivalent to approximately 9–14 decades of current global, annual, human-induced carbon emissions. Approximately, eight trillion tons of water evaporate from Amazon forests each year, with important influences on global atmospheric circulation (Nepstad et al., 2008).

Savannas are the most extensive biome in the tropics, and important spatial extensions in the subtropic, that has been shaped by a long history of interaction with humans, fire, climate and wildlife. The impacts on savanna composition, distribution and function based on increasing human population growth, climate change, atmospheric change and resource use impact, bring multidimentional challenges, within the political realms, land tenures and economic shifts, what in fact requires effective long-term management strategies and thus ensure a sustainable future for savanna ecosystems (Marchant, 2010).

The neotropical Atlantic Forest supports one of the highest degrees of species richness and rates of endemism on the planet, but has also undergone a huge forest loss, for example the Brazilian Atlantic Forest is highly fragmented and with just 12–16% of the original forest cover left (Ribeiro et al., 2009).

There are differences in state of knowledge of the marine biodiversity among the subregions, and even though incomplete in some areas, there are differences in total biodiversity among Atlantic and Pacific oceans at the same latitude. At north of the continent, the Tropical East Pacific is richer in species than the Tropical West Atlantic. In the south, the Humboldt Current system is much richer than the Patagonian Shelf. An analysis of endemism shows that 75% of the species are reported within only one of the South America regions, while about 22% of the species of South America are not reported elsewhere in the world (Miloslavich et al., 2011).

Historical note and biomes transformation in the Americas

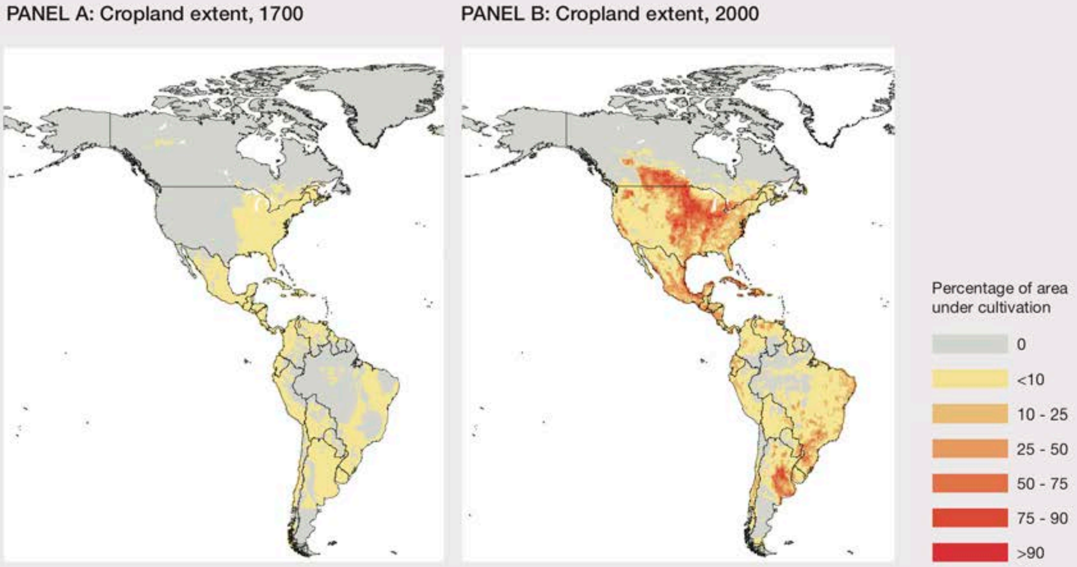
The region is populated by a uniquely large proportion of new or descendants of immigrants from all parts of Europe, Asia and Africa, in addition to over 66.2 million indigenous peoples who have persisted culturally despite centuries of land expropriation and in some cases active persecution and genocide (Chapter 2). All subregions have had the representation of units of analysis extensively altered post 1500, when immigration from the Old World and subsequent expansion of European style “settlement” brought new cultures and more advanced technologies and to the Americas. These contrasts may be particularly informative for development of effective policies, by shedding light on how socio-economic factors affect conservation policies and measures, and how economic and social policies perform in different biotic settings.

The Americas have experienced extensive change in biomes, with notable expansion of croplands in the last three centuries (Figure 1.9). The origin of crops, and their precursors, and current growing location of crops go hand in hand. The ‘centers of origin’ of crops are a theme of considerable debate (Beddow et al., 2010). However, there is little doubt that the process of domestication and geographical dispersal are part of the broader history of human-induced spatial movement of plants and animals. Candolle (1884) observed that ancient plant propagation in the Mediterranean by the Egyptians and Phoenicians enabled subsequent migrants to carry West Asian genetic material to Europe at least 4,000 years ago; there is well-established Chinese cultivation of rice, sweet potatoes, wheat and millets as early as 2,700 BC.

The rate at which human action has driven development, improvement and movement of plants and animals has accelerated significantly in the past 500 years (Beddow et al., 2010). The “Colombian Exchange” was an important historical events initiated when Columbus made contact with Native Americans in the New World (Crosby, 1987; Diamond, 1999). Beddow et al. (2010) emphasize that “most of the commercial agriculture in the USAtoday is based on crop and livestock species introduced from Eurasia (e.g. wheat, barley, rice, soybeans, grapes, apples, citrus, cattle, sheep, hogs, and chickens), though with significant involvement of American species (e.g. corn, peppers, potatoes, tobacco, tomatoes, and turkeys) that are also distributed throughout the rest of the world.” The global movement of agriculturally important plants and animals, and their accompanying pests and diseases, has been a pivotal element in both the history of agriculture and transformation of biomes in the Americas.

Figure 1 9 **The changing global landscape of crop production.**

Panels A and B illustrate the extent of crop production in the Americas in 1700 and 2000. Areas with darker shades, as in Panel B, are devoted to more intensive cropping. Source: modified from Beddow *et al.* (2010) derived from SAGE data.



1.6.2 Cultural aspects: Presence of indigenous groups, population, and land holdings

There are at least 66 million indigenous people in the four subregions of the Americas, ranging from 89.29% of indigenous people in Greenland to 0.04% in Cuba (Tables 1.3). However, the percentage of the indigenous population in each country, sourced from either official censuses or other surveys, could be higher than values presented in the tables below. There are some countries, for example, where more than half of the indigenous population live in urban areas - such as Mexico, Peru, Uruguay and Venezuela - and are not captured in these statistics. Self-declaration is also another cause of under-representation in census data of the Americas. For example, the Amazon region alone has outstanding cultural diversity with 420 indigenous and tribal peoples, 86 languages and 650 dialects (www.otca.info) and wealth of ILK (Berkes, 2012; Tengö et al., 2014), but faces poverty and social inequality (PNUD, 2013; Ioris, 2016).

The results in the tables below show the information gap, especially among the Caribbean countries, where there are almost no records or quantitative data. This does not imply the absence of indigenous groups or land in a given country. In the broader Caribbean region the indigenous populations were almost totally decimated by colonization in the post-Columbus era. To find evidence of indigenous groups' population and territorial holdings in these countries required the use of other sources of information. A considerable amount of information for this subsection was found in magazines of local and other international organizations, such as “Cultural Survival”.

Table 1.3. Indigenous population (IP) in the Americas

Region	Country	*1000 (thousands)		% IP/PC	Year
		Population Country (PC)	Indigenous population (IP)		
North America		357,327	8,051	2,3	
	Greenland	56[a]	50[b]	89,3	2017
	Canada	35,852[a]	1,401[b]	3,9	2017
	USA	321,419[a]	6,600[b]	2,1	2017
Mesoamerica		172,740	33,778	19,6	
	Mexico	127,017[a]	21,497[j]	13,3	2015
	Guatemala	16,343[a]	9,805[b]	60,0	2017
	Nicaragua	6,082[a]	567[b]	9,3	2017
	Costa Rica	4,808[a]	104[c]	2,2	2010
	Panama	3,929[a]	418[c]	10,6	2010
	Honduras	8,075[a]	922[d]	11,4	2006
	Belize	359[a]	44[d]	12,3	2006
	El Salvador	6,127[a]	422[d]	6,9	2006
South America		418,420	24,277	5,8	
	Argentina	43,416[a]	955[b]	2,2	2017
	Bolivia	10,725[a]	5,652[d]	52,7	2006
	Brazil	207,848[a]	897[e]	0,4	2010
	Chile	17,948[a]	1,566[f]	8,7	2013
	Colombia	48,229[a]	1,500[b]	3,1	2016
	Ecuador	16,144[a]	1,018[c]	6,3	2010
	Guyana	767[a]	51[d]	6,6	2006
	French Guyana	244[b]	10[b]	4,1	2017
	Paraguay	6,639[a]	113[b]	1,7	2017
	Peru	31,377[a]	11,655[d]	37,1	2006
	Surinam	543[a]	20[b]	3,7	2017
	Uruguay	3,432[a]	115[g]	3,4	2004
	Venezuela	31,108[a]	725[c]	2,3	2010
Caribbean		38,009			
	Antigua and Barbuda	92[a]			
	The Bahamas	388[a]	3[d]	0,8	2006
	Barbados	284[a]			
	Cuba	11,390[a]	5[h]	0,0	2011
	Dominica	73[a]	3[i]	4,1	2017
	Grenada	107[a]			
	Haiti	10,711[a]			
	Jamaica	2,726[a]	51[d]	1,9	2006
	Dominican Republic	10,528[a]			
	St. Lucia	185[a]			
	St. Kitts and Nevis	56[a]			
	St. Vincent and the Grenadines	109[a]			
	Trinidad and Tobago	1,360[a]	26[d]	1,9	2006

a. World Bank (2015)
b. Hansen *et al*. (2017)
c. CEPAL (2010)
d. Montenegro & Stephens (2006)
e. Instituto Socioambiental (ISA) (2010)
f. Ministerio de Desarrollo Social de Chile (2013)
g. Lopez (2009)
h. Poole (2011)
i. Kalinago (2017)
j. Instituto Nacional de Estadística y Geografia México (2015)

There is an area of around 272 million hectares of indigenous lands in different countries of the Americas (Table 1.4). One initial criteria include the presence or extension of indigenous people lands legally recognized in constitutional country-based legislations and/or international agreements such as Convention 169 of the International Labor Organization. However, although countries like Chile are signatories of this international convention, laws in this country do not recognize “land property” owned by indigenous communities. In other cases, there is no legal land recognised at the community level as in Trinidad and Tobago and Suriname.

Table 1.4. Indigenous land in the Americas

Region	Country	*1000 (ha)		%
		Country Area[a]	Indigenous land	Indigenous land/Country Area
North America		2198,227	25,500	1,2
	Greenland	216,609[b]		
	Canada	998,467	2,800[c]	0,3
	USA	983,151	22,700[d]	2,3
Mesoamerica		248,676	48,495	19,5
	Mexico	196,438	45,700[e]	23,3
	Guatemala	10,899	1,531 [e]	14,0
	Nicaragua	13,037		
	Costa Rica	5,110	334[f]	6,5
	Panama	7,542	753 [e]	10,0
	Honduras	11,249	160 [e]	1,4
	Belize	2,297	17[g]	0,7
	El Salvador	2,104		
South America		1780,326	197,813	11,1
	Argentina	279,181	Nd	
	Bolivia	109,858	20,000[f]	18,2
	Brazil	851,577	117,310 [h]	13,8
	Chile	75,610	328[i]	0,4
	Colombia	114,175	36,337[J]	31,8
	Ecuador	25,637	6,830[e]	26,6
	Guyana	21,497	3,108[k]	14,5
	French Guyana	8,385	Nd	
	Paraguay	40,675		
	Peru	128,522	13,200 [e]	10,3
	Surinam	16,382	0	0
	Uruguay	17,622		
	Venezuela	91,205	700 [e]	0,8
Caribbean				
	Antigua and Barbuda	44		44
	The Bahamas	1,388		1,388
	Barbados	43		43
	Cuba	10,989		10,989
	Dominica	75	2[l]	75
	Grenada	35		35
	Haiti	2,775		2,775
	Jamaica	1,099		1,099
	Dominican Republic	4,867		4,867
	St. Lucia	62		62
	St. Kitts and Nevis	26		26

	St. Vincent and the Grenadines	39	0,1[m]	39
	Trinidad and Tobago	513	0[n]	513
	a. IBGE (2017) b. Central Intelligence Agency (2015) c. Statistics Canada (201) d. USA Department of the Interior Indians Affair (2017) e. Blaser *et al.* (2011) f. Hansen *et al.* (2017) g. Cultural Survival Quarterly Magazine 82013) Nd: No data		h. Instituto Socioambiental (ISA) (2017) i. FAO (2012b) j. Van Dam (2011) k. Amerindian Peoples Association (2017) l. Kalinago Territory (2017) m. Cultural Survival Quarterly Magazine (2017) n. Santa Rosa First Peoples Community (2015)	

1.6.3 Socio-economic features

The population in the Americas represent 15% of the total global human population (UNDP, 2016) with a population density in the Americas ranges from 2 per 100 km^2 of land in Greenland to over 9,000 per km^2 in several core urban centers. It includes the most urbanized regions in the world (North America and Latin America and the Caribbean with 82% and 80% of inhabitants living in urban areas respectively) (UN-DESA, 2016). Five cities in the Americas (Sao Paulo, Mexico DF, New York-Newark, Buenos Aires and Rio de Janeiro) are in the top 20 world's megacities (more than 10 million inhabitants) in 2016 (UN-DESA, 2016).

Patterns of economic growth differ both, among and within the subregions. Some key socio-economic indicators such as the GDP[4], the Globalization index[5] or the HDI[6] show marked differences between subregions (Figure 1.10). There is a clear contrast between North American countries and the rest of the region. South America presents a high heterogeneity in the three indicators. The Americas contains two of the top 10 countries with the highest HDI as well as one of the countries with lowest human development (UNDP, 2016). Economic growth and international trade have improved the quality of life of many people, but often at the cost of increasing demand for natural resources, which affect other group's quality of life. Overall, poverty levels have decreased in the last two decades but groups in Mesoamerica, the Caribbean and South America are yet facing poverty (Chapter 2). Such heterogeneity hampers developing general conclusions that apply equally across all subregions.

[4] GDP at purchaser's prices is the sum of gross value added by all resident producers in the economy plus any product taxes and minus any subsidies not included in the value of the products. It is calculated without making deductions for depreciation of fabricated assets or for depletion and degradation of natural resources. Data are in current USA dollars. Dollar figures for GDP are converted from domestic currencies using single year official exchange rates. For a few countries where the official exchange rate does not reflect the rate effectively applied to actual foreign exchange transactions, an alternative conversion factor is used (World Bank, 2017).

[5] The index of globalization covers three main dimensions: economic integration, social integration, and political integration. Using panel data for 123 countries in 1970-2000 it is analyzed empirically whether the overall index of globalization as well as sub-indexes constructed to measure the single dimensions affect economic growth. As the results show, globalization indeed promotes growth. The dimensions most robustly related with growth refer to actual economic flows and restrictions in developed countries. Although less robustly, information flows also promote growth whereas political integration has no effect (Gygli et al., 2018).

[6] Human Development Index (HDI) is a composite index constructed by combining a range of indicators that aim at capturing human achievement in three dimensions: per capita income, education, and life expectancy (UNDP, 2016).

Figure 1 10 **Gross Domestic Product, Globalization and Human Development levels in countries of the Americas. Source: World Bank (2017), Gygli *et al.* (2018) and UNDP (2016).**

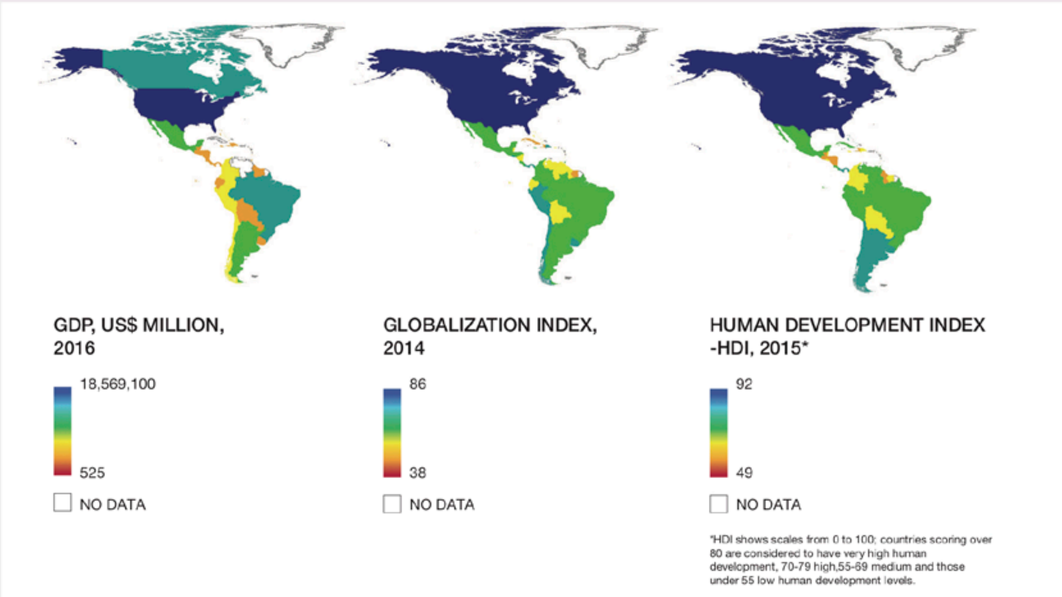


Table 1.5 presents several subregional socio-economic indicators with their average values by indicator, along with the lowest and highest value across the states. Because the countries differ in size as well as development, indicators that are national totals rather than per capita values should be compared with caution. Even within countries some socio-economic factors like personal income have such skewed distributions that an average value may represent status of the citizenry very poorly.

Table 1.5. Socio-economic indicators by subregion

Descriptors	North America	Mesoamerica	South America	Caribbean	Source
Countries included in the assessment	Canada, USA, Greenland	Belize, Costa Rica, El Salvador, Guatemala, Honduras, Mexico, Nicaragua and Panama	Argentina, Bolivia (Plurinational State of), Brazil, Chile, Colombia, Ecuador, Guyana, Paraguay, Peru, Suriname, Uruguay and Venezuela (Bolivarian Republic of)	Antigua and Barbuda, The Bahamas, Barbados, Cuba, Dominica, Dominican Republican[7], Grenada, Haiti, Jamaica, Saint Kitts and Nevis, Saint Lucia, Saint Vincent and the Grenadines and Trinidad and Tobago	IPBES (2015a)
Total area (km^2): 41,858,533	21,415,862	2,477,901	17,730,93	233,839	
Social and demographic indicators					
Population (million inhabitants, 2015)	~360	~ 175	~ 418	~ 38	World Bank (2015)
Adult literacy rate 15+ years (%), 2015 Mean (min-max)	84% (USA) – 99% (Canada)	88.5 (79-98)	95 (88 – 98)	88 (61-100) Data available for 5 countries	World Bank (2015) (National statistics)
Industry, value added (% of GDP), 2014 Mean (min-max)	20.7 Data only available for USA	26.3 (18-32)	32.2 (21-42)	21.6 (11.3-48.8) Data not available for Haiti	World Bank (2015)
Gross National Income per capita (US dollars, 2013 for South America and 2015 for the rest of subregions) Mean (min-max)	51,615 (47,250-55,980) Data not available for Greenland	6,028 (1940-11880)	8,954 (2,620 – 15,580)	10,219 (810-20,740) Data not available for Cuba	World Bank (2015)

1.6.4 Governance in the Americas

For this Assessment "governance" will be discussed in several chapters, referring to structures and processes that are designed to ensure accountability, transparency, responsiveness, rule of law, stability, equity and inclusiveness, empowerment, and broad-based participation. Governance is more than the institutions of the government, but encompasses all the ways that social units of people are structured and

[7] On socioeconomic, cultural and historical grounds, the Dominican Republic could be considered part of Mesoamerica, and Guyana part of the Caribbean.

managed to meet a need or to pursue collective goals (UNESCO, 2017). In the Americas many different types of governance arrangements have developed. These occur in different social, economic and environmental contexts, associated with a diverse range of institutional arrangements and mechanisms that operate at multiple scales of intervention.

The IPBES Assessment does not analyse governance structures and mechanisms. However, since governance reflects the norms, values and rules through which public affairs are managed and includes the culture and institutional environment in which citizens and stakeholders interact among themselves and participate in public affairs, it is relevant to explaining many of the patterns and trends discussed throughout the Assessment. It is also a relevant consideration in contemplating potential pathways and policy options for the future. Consequently, some higher level features of governance in the subregions are summarized below (Table 1.6).

In terms of governance, the single greatest difference among subregions may be simply in the size and number of independent States, with North America, the geographically largest subregion, comprised of only Canada, the USA, and Greenland (under Danish rule). The geographically smallest region, the Caribbean, on the other hand, includes 13 independent States and 13 Protectorates. The indicators of Governance are taken from the Worldwide Governance Indicators (Kaufmann et al., 2010) and The Economist Group (http://www.economistgroup.com/) to provide some insight into the degree to which governance processes can support efforts to conserve and sustain biodiversity and maintain deliver of NCP.

Table 1.6. Governance indicators by subregion

Descriptors[8]	North America*	Mesoamerica	South America	Caribbean*	Source
Political Instability Index, 2009-2010	4.05 (2.8-5.3) Data not available for Greenland	5.9 (3.5 -7.1)	6.6 (5.1-7.7)	6.06 (4.2-7.8) Data available for 5 countries	The Economist Group[9]
Political Stability and Absence of Violence or Terrorism (Percentile Rank 0-100), 2015	88 (70-100)	42.12 (18-64)	41.2 (12 – 83)	68.76 (22-97)	Kaufmann *et al.* (2010)
Rule of law (0-100 rank), 2015	92 (90-95)	34.1 (15-69)	41 (11-87)	56 (10-82)	Kaufmann *et al.* (2010)
Control of corruption (0-100 rank), 2015	89 (84-94)	40 (19-75)	39 (6-89)	59 (9-93)	Kaufmann *et al.* (2010)

* Greenland and the 13 Caribbean Protectorates are still colonies of European States, so their governance aspects are not included in this table

[8] The Political Instability Index shows the level of threat posed to governments by social protest. The index scores are derived by combining measures of economic distress and underlying vulnerability to unrest.
The Political Stability and Absence of Violence/Terrorism index captures perceptions of the likelihood that the government will be destabilized or overthrown by unconstitutional or violent means, including politically-motivated violence and terrorism.
The Rule of Law index captures perceptions of the extent to which agents have confidence in and abide by the rules of society, and in particular the quality of contract enforcement, property rights, the police, and the courts, as well as the likelihood of crime and violence.
The Control of Corruption index captures perceptions of the extent to which public power is exercised for private gain, including both petty and grand forms of corruption, as well as "capture" of the state by elites and private interests.
The Worldwide Governance Indicators are available at: www.govindicators.org

[9] http://viewswire.eiu.com

1.7 Technical details: Methods and approaches in the Assessment

1.7.1 How this Regional Assessment deals with incomplete or absent information

An assessment on a continental scale is built on the basis of numerous sources of information. Although there is immense value in an assessment that can incorporate many sources of information, there are also many challenges to overcome, including incomplete or absent information, low quality information, limits in representativeness of information sources. To address these issues consistently, this Assessment follows the guidelines provided by the IPBES Task Force on Knowledge and Data. The identification and classification of gaps in knowledge are necessary contributions to support decisions, conservation and for ongoing and future assessment processes.

The collection, processing and use of data, information and knowledge followed certain key principles and practices to meet quality standards to ensure that the target audiences have sufficient confidence in the Assessment conclusions to use them in policy and decision-making. Among these principles and practices are: i) inclusion of all relevant and available or readily mobilizable data, information and knowledge from different knowledge systems and sources; ii) transparency at all steps of collection, selection, analysis and archiving, in order to enable informed feedback on Assessments and replicability of results, and to enable comparability across scales and time; and iii) systematic and well-documented methodology in all steps of the assessment process, including documentation of the representativeness of the available evidence, the remaining gaps and uncertainty, and iv) clear rationales in cases where a "weight of evidence" conclusion was drawn from the broad range of relevant information presented in i).

1.7.2 How this Regional Assessment handles uncertainty

Uncertainty in assessments arises from several sources, including the incompleteness or unrepresentativeness of information available; having information available that is of low accuracy, precision or both (whether accuracy and precision have been estimated or not); and having multiple studies that individually may report finding of moderate accuracy and precision, but are inconsistent with each other across studies. In the case of uncertainty, each chapter of this report establishes the level of confidence in relation to the key findings (data and information from the ensemble of knowledge systems) presented in Executive summaries. Each key finding in an IPBES Assessment comes with a confidence language statement. In Assessments, when we talk about confidence in relation to knowledge, we are referring to how assured the experts are about the findings presented within their chapters. Low confidence describes a situation where we have incomplete knowledge and therefore cannot fully explain an outcome or reliably predict a future outcome, whereas high confidence conveys that we have extensive knowledge and are able to explain an outcome or predict a future outcome with much greater certainty.

The communication of confidence in IPBES Assessments is important because interactions between humans and the natural world are complex, as are the interactions among people relative to nature. To allow decision makers to make informed decisions, author teams need to communicate not only the findings in which they have high confidence but also those in which their confidence is weaker, in cases when the finding is the best inference that can be drawn from the knowledge available. Furthermore, by following a common approach to applying confidence terminology within an Assessment, authors are able to increase consistency and transparency.

IPBES Assessments uses four specific phrases known as "confidence terms" in order to categorize the experts' level of confidence in their findings consistently (Figure 1.11). The categories depend on the author team's expert judgment on the quantity and quality of the supporting evidence and the level of scientific agreement about what that evidence shows. The Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services Assessments use a four-box model of confidence (below) based on evidence and agreement that gives four main confidence terms: "well established" (much evidence and high agreement), "unresolved" (much evidence but low agreement), "established but incomplete" (limited evidence but good agreement) and "inconclusive" (limited or no evidence and little agreement).

Figure 1 11 The four-box model for the qualitative communication of confidence.

Confidence increases towards the top-right corner as suggested by the increasing strength of shading. Source: IPBES (2016).

Depending on the nature of the evidence supporting the key message or finding, quantitative assessments of confidence may also be possible. Quantitative assessments of confidence are estimates of the likelihood (probability) that a well-defined outcome will occur in the future. Probabilistic estimates are based on statistical analysis of observations or model results, or both, combined with expert judgment. However, it may be that quantitative assessments of confidence are not possible in all assessments, due to limitations in the evidence available.

1.7.3 Data and indicators

The Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services uses indicators in conducting its Assessments. Indicators are defined here as data aggregated in a quantitative or qualitative manner that reflect the status, cause or outcome of an object or process, especially towards targets such as the Aichi targets or those set by the SDG. Indicators can help simplify the enormous complexity of datasets, variables, frameworks and approaches available to us. They are also useful tools for communicating the results of assessments. It is, however, important to recognize the limitations of a given set of indicators in capturing the complexities of the 'real world', since indicators are restricted to what can be measured in a standardized way and for which appropriate data are widely available with good global coverage. Notably, these limitations are especially significant when it comes to assessing non-material benefits of nature to people and in quality of life. Moreover, the meanings of indicators are related to diverse cultural perspectives. Hence, in IPBES Assessments, indicators are subjected to critical analysis and review from a diversity of experts.

The Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services has consulted widely in arriving at a list of 30 indicators for its Assessments, of which nine are intended to assess socio-ecological status and trends. Indicators have been selected to cover the Conceptual Framework comprehensively as well as being interpretable in what relates to drivers, pressure, status, impact, response´s approach to assessments. Table 1.7 lists the indicators with their role related to drivers, pressure, status, impact, response and IPBES conceptual framework, and their sources in other agencies or more thematically focused assessments.

Table 1.7. Core and socio-economic indicators used in IPBES assessments

Specific Indicator	Aichi target	DPSIR*	CF**	Source
Core indicators				
Ecological Footprint	4	P	DD	Global Footprint Network
Water Footprint	4	P	DD	Water Footprint Network
Percentage of Category 1 nations in CITES	4	R	IGID	Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES)
Biodiversity Habitat Index	5	S	DD, BEF	GEO BON – CSIRO
Species Habitat Index	5, 12	P,S	DD, BEF	GEO BON - Map of Life
Forest area as a percentage of total land area	5	S	DD, BEF	FAO
Trends in forest extent (tree cover)	5	S	DD, BEF	Hansen et al., 2013
Protected area coverage of Key Biodiversity Areas (including Important Bird and Biodiversity Areas, Alliance for Zero Extinction sites)	5, 11, 12	R	IGID, DD	BirdLife International, the International Union for Conservation of Nature (IUCN), UNEP-WCMC
Total wood removals	5, 7, 14	S,I	DD, NBP	FAO
Estimated fisheries catch and fishing effort	6	P	DD, BEF	Sea Around Us
Proportion of fish stocks within biologically sustainable levels	6	S	BEF	FAO
Inland fishery production	6, 14	S, I	BEF, NBP	FAO
Marine Trophic Index	6	S	DD, BEF	Sea Around Us
Trends in fisheries certified by the Marine Stewardship Council	6	R	IGID	Marine Stewardship Council
Proportion of area of forest production under FSC and PEFC certification	7	R	IGID, DD	Forest Stewardship Council (FSC), Programme for the Endorsement of Forest Certification (PEFC)
Nitrogen Use Efficiency	7	P	DD	Lassaletta et al., 2014 from Environmental Performance Index (EPI)
Nitrogen + Phosphate Fertilizers (N+P205 total nutrients)	7	P	DD	FAO
Trends in pesticide use	8	P	DD	FAO
Trends in nitrogen deposition	8	P	DD	International Nitrogen Initiative
Protected Area Connectedness Index	11	R	DD, IGID	GEO BON – CSIRO
Percentage of areas covered by protected areas - marine, coastal, terrestrial, inland water	11	R	IGID	UNEP-WCMC, IUCN
Species Protection Index	11	P,R	IGID, DD	GEO BON - Map of Life
Protected area management effectiveness	11	R	IGID, DD, BEF	UNEP-WCMC
Biodiversity Intactness Index	12, 14	P,S	DD, BEF	GEO BON – PREDICTS

Red List Index	12	S	BEF	IUCN, BirdLife International and other Red List Partners
Proportion of local breeds, classified as being at risk, not-at-risk or unknown level of risk of extinction	13	S	BEF, NBP	FAO
Percentage of undernourished people	14	I	GQL	FAO
Number of countries with developed or revised NBSAPs	17	R	IGID	Secretariat of the Convention on Biological Diversity (CBD)
Proportion of known species assessed through the IUCN Red List	19	R	IGID	IUCN
Species Status Information Index	19	R	IGID, BEF	GEO BON - Map of Life

Specific Indicator	DPSIR*	CF**	Source
Socio-economic indicators			
GDP	S	IGID	World Bank
Food security: Countries requiring external assistance for food (famine relief)	S	GQL	FAO
Food security: Calorie supply per capita (kcal/capita.day)	S	GQL	FAO
Water security: Proportion of population using safely managed drinking water services (SDG 6.1.1)	S	GQL	UNICEF/WHO
Water security: Freshwater consumption as % of total renewable water resources	S	GQL	FAO
Equity: GINI index	S	GQL	World Bank
Food: World grain production by type/capita.year	S	NCP	FAO
Non-material NCP: Index of Linguistic Diversity (ILD)	S,P	NCP, IGID	UNESCO

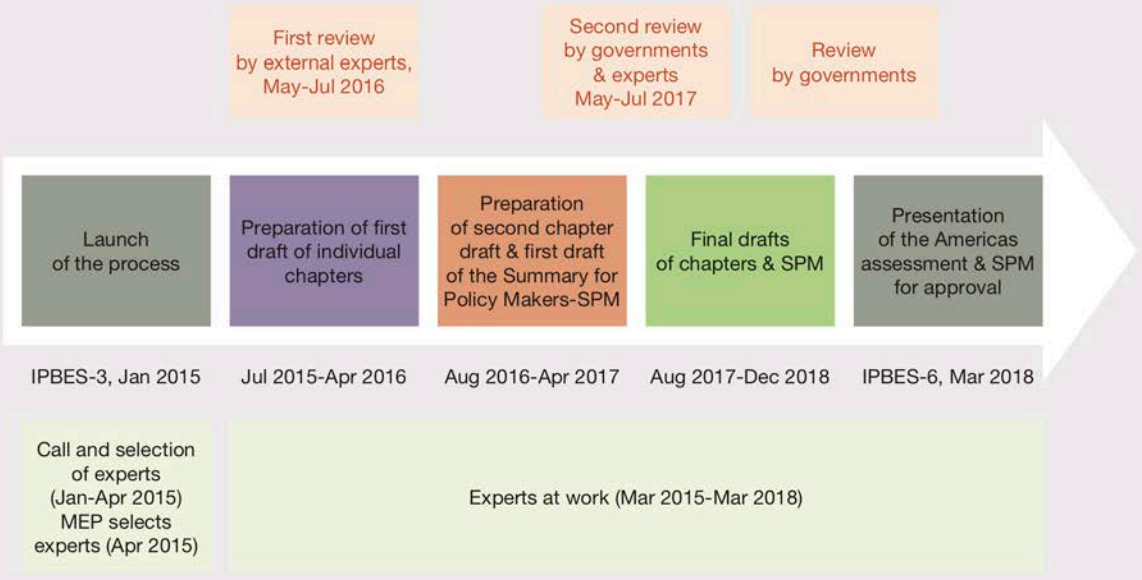
* DPSIR – D: Drivers, P: Pressure, S: Status, I: Impact, R: Response

** CF (Conceptual Framework) – DD: direct driver, NCP: nature's contributions to people/ ecosystem goods and services, /biodiversity and ecosystem functions, IGID: institutions, governance and other indirect drivers, GQL: good quality of life/human well-being

Source: IPBES (2017b) and https://www.ipbes.net/indicators/socioeconomic

1.7.4 Process for the production of the Americas assessment report

Figure 1 12 Process for the production of the Americas assessment report. Source: own representation.



This Assessment Report is the result of a four-year process containing five phases (Figure 1.12) and involving more than one hundred experts. At the beginning of 2015 - during the IPBES-3 Plenary - the scope, geographic area, rationale, utility and assumptions of this Assessment were agreed and approved. Then the process of call and selection of experts (until April 2015) resulted in 92 experts from 20 countries. In addition, through the Technical Support Unit Capacity Building, a pilot program for young researchers was carried out and 6 fellows were selected throughout the continent (one fellow for each chapter).

During March 2015 to March 2018 the experts worked on the elaboration of this Report, which encompassed the preparation of two drafts (which were submitted for external review by experts and governments). After the second draft, a selection of experts working on the Regional Assessment also worked in the construction of the summaries for policymakers. The process will conclude with the presentation of the Americas Assessment and Summary for Policy Makers for approval by the sixth session of the IPBES Plenary (IPBES-6) held in Medellin, Colombia in March 2018.

1.8 References

Aide, T. M., & Grau, H. R. 2004). Globalization, migration, and Latin American ecosystems. *Science*, 305(5692), 1915-1916.

Altieri, M. A., & Toledo, V. M. (2011). The agroecological revolution in Latin America: rescuing nature, ensuring food sovereignty and empowering peasants. *Journal of Peasant Studies*, 38(3), 587-612.

Alvarez-Filip, L., Dulvy, N. K., Gill, J. A., Côté, I. M., & Watkinson, A. R. (2009). Flattening of Caribbean coral reefs: region-wide declines in architectural complexity. *Proceedings of the Royal Society of London B: Biological Sciences*, 276(1669), 3019-3025.

Amerindian Peoples Association (2017). Land and Territorial Rights. Retrieved from http://apaguyana.weebly.com/land-and-territorial-rights.html.

Anderson-Teixeira, K. J., Snyder, P. K., Twine, T. E., Cuadra, S. V., Costa, M. H., & DeLucia, E. H. (2012). Climate-regulation services of natural and agricultural ecoregions of the Americas. *Nature Climate Change*, *2*(3), 177–181.

Arriagada, R. A., Ferraro, P. J., Sills, E. O., Pattanayak, S. K., & Cordero-Sancho, S. (2012). Do Payments for Environmental Services Affect Forest Cover? A Farm-Level Evaluation from Costa Rica. *Land Economics*, *88*(2), 382–399.

Ash, N., Blanco, H., Brown, C., Garcia, K., Henrichs, T., Lucas, N., Ruadsepp-Heane, C., Simpson, R. D., Scholes, R., Tomich, T., Vira, B., & Zurek, M. (2010). *Ecosystems and human well-being : a manual for assessment practitioners.* Washington DC: Island Press. Retrieved from http://www.ecosystemassessments.net/resources/ecosystems-and-human-well-being-a-manual-for-assessment-practitioners.pdf

Barral, M. P., Rey Benayas, J. M., Meli, P., & Maceira, N. O. (2015). Quantifying the impacts of ecological restoration on biodiversity and ecosystem services in agroecosystems: A global meta-analysis. Agriculture, Ecosystems & Environment, 202, 223–231.

Baylis, K., Honey-Rosés, J., Börner, J., Corbera, E., Ezzine-de-Blas, D., Ferraro, P. J., Lapeyre, R., Persson, U. M., Pfaff, A., & Wunder, S. (2016). Mainstreaming Impact Evaluation in Nature Conservation. *Conservation Letters*, *9*(1), 58–64.

Bazile, D., Jacobsen, S.-E., & Verniau, A. (2016). The Global Expansion of Quinoa: Trends and Limits. *Frontiers in Plant Science*, *7*, 622.

Beard, J. S. (1944). Climax Vegetation in Tropical America. *Ecology*, *25*(2), 127–158.

Beddow, J. M., Pardey, P. G., Koo, J., & Wood, S. (2010). The changing landscape of global agriculture. In J. M. Alston, B. A. Babcock, & P. G. Pardey (Eds.), The Midwest Agribusiness Trade Research and Information Center Iowa State University. Ames, Iowa.

Bennett, E. M., Lazos, E., Woodward, G., Bennett, E. M., Cramer, W., Begossi, A., Egoh, B. N., Cundill, G., Dı, S., Geijzendorffer, I. R., Krug, C. B., Lavorel, S., Meyfroidt, P., Mooney, H. A., Nel, J. L., Pascual, U., & Payet, K. (2015). Linking biodiversity, ecosystem services , and human well-being : three challenges for designing research for sustainability ScienceDirect Linking biodiversity, ecosystem services, and human Anne-He, (JUNE), 76–85. https://doi.org/10.1016/j.cosust.2015.03.007

Bennett, E. M., Peterson, G. D., & Gordon, L. J. (2009). Understanding relationships among multiple ecosystem services. *Ecology Letters*, 12(12), 1394-1404.

Bennett, G., Gallant, M., & Ten Kate, K. (2017). *State of Biodiversity Mitigation 2017: Markets and Compensation for Global Infrastructure Development*. Washington, D.C. https://doi.org/10.18235/0000675

Berbés-Blázquez, M., González, J. A., & Pascual, U. (2016). Towards an ecosystem services approach that addresses social power relations. *Current Opinion in Environmental Sustainability*, 19, 134-143.

Berkes, F. (2012). Sacred ecology (Third). New York and London: Routledge Taylor & Francis Group.

Bhaduri, A., Bogardi, J., Siddiqi, A., Voigt, H., Vörösmarty, C., Pahl-Wostl, C., Bunn, S. E., Shrivastava, P., Lawford, R., Foster, S., Kremer, H., Renaud, F. G., Bruns, A., & Rodríguez Osuna, V. (2016). Achieving Sustainable Development Goals from a Water Perspective. *Frontiers in Environmental Science*, *4* (64), 1–13.

Blaser, J., Sarre, A., Poore, D., & Johnson, S. (2011). *Status of Tropical Forest Management report released | The International Tropical Timber Organization (ITTO)*. Yokohama. Retrieved from http://www.itto.int/news_releases/id=2663

Boschetti, F., Walker, I., & Price, J. (2016). Modelling and attitudes towards the future. *Ecological Modelling*, *322*, 71–81.

Bruckner T., I.A. Bashmakov, Y. Mulugetta, H. Chum, A. de la Vega Navarro, J. Edmonds, A. Faaij, B. Fungtammasan, A. Garg, E. Hertwich, D. Honnery, D. Infield, M. Kainuma, S. Khennas, S. Kim, H.B. Nimir, K. Riahi, N. Strachan, R. Wiser, and X. Zhang. (2014). *Energy Systems*. In: *Climate Change 2014: Mitigation of Climate Change. Contribution of Working Group III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change* [Edenhofer, O., R. Pichs-Madruga, Y. Sokona, E. Farahani, S. Kadner, K. Seyboth, A. Adler, I. Baum, S. Brunner, P. Eickemeier, B. Kriemann, J. Savolainen, S. Schlömer, C. von Stechow, T. Zwickel and J.C. Minx (eds.)]. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Buonocore, J. J., Luckow, P., Norris, G., Spengler, J. D., Biewald, B., Fisher, J., & Levy, J. I. (2015). Health and climate benefits of different energy-efficiency and renewable energy choices. *Nature Climate Change*, 6(1), 100–105.

Buytaert, W., & De Bièvre, B. (2012). Water for cities: The impact of climate change and demographic growth in the tropical Andes. *Water Resources Research*, 48(8), 1–13.

Campetella, C. M., & Vera, C. S. (2002). The influence of the Andes mountains on the South American low-level flow. *Geophysical Research Letters*, 29(17).

Candolle, A. (1884). *Origin of cultivated plants (The International scientific series)* (49th ed.). London: Kegan Paul, Trench and Co. Retrieved from https://www.amazon.com/Origin-cultivated-plants-International-scientific/dp/B00088N86M

Carpenter, S., Bennett, E., & Peterson, G. (2006). Scenarios for ecosystem services: an overview. *Ecology and Society*, 11(1), 29.

CBD. (2010). Linking Biodiversity Conservation and Poverty Aleviation: A State of Knowledge Review. Montreal, Quebec.

CBD/FAO/WB/UNEP/UNDP. (2016). Biodiversity and the 2030 Agenda for Sustainable Development. Montreal, Quebec: Convention on Biological Diversity. Retrieved from http://www.undp.org/content/undp/en/home/librarypage/environment-energy/ecosystems_and_biodiversity/biodiversity-and-the-2030-agenda-for-sustainable-development---p.html

Central Intelligence Agency. (2015). World Bank Open Data. Retrieved from https://www.cia.gov/library/publications/resources/the-world-factbook/geos/gl.html

CEPAL. (2010). Los pueblos indigenas em América Latina. Sistemas de Indicadores Sociodemográficos de Poblaciones Y Pueblos Indígenas. Comisión Económica para América Latina y el Caribe. Retrieved from http://celade.cepal.org/redatam/PRYESP/SISPPI/.

Chan, K. M. A., Balvanera, P., Benessaiah, K., Chapman, M., Díaz, S., Gómez-Baggethun, E., Gould, R., Hannahs, N., Jax, K., Klain, S., Luck, G. W., Martín-López, B., Muraca, B., Norton, B., Ott, K., Pascual, U., Satterfield, T., Tadaki, M., Taggart, J., & Turner, N. (2016b). Why protect nature? Rethinking values and the environment. *PNAS*, *113*(6), 1462–1465.

Chivian E. and A. Bernstein. (2010). How our Health depends on Biodiversity. Center for Health and the Global Environment. Boston.

CIP. (2017). Potato Facts and Figures. A CGIAR Research Center. International Potato Center. Retrieved from http://cipotato.org/

Cisneros, E., Zhou, S. L., & Börner, J. (2015). Naming and Shaming for Conservation: Evidence from the Brazilian Amazon. *PLOS ONE*, 10(9), e0136402.

Clamen, M., & Macfarlane, D. (2015). The International Joint Commission, Water Levels, and Transboundary Governance in the Great Lakes. Review of Policy Research, 32(1), 40–59.

Claudio, L., Stingone, J., & Godbold, J. (2006). Prevalence of Childhood Asthma in Urban Communities: The Impact of Ethnicity and Income. *Annals of Epidemiology*, 16(5), 332–340.

Clement, C. R., de Cristo-Araújo, M., d'Eeckenbrugge, G. C., Alves Pereira, A., & Picanço-Rodrigues, D. (2010). Origin and Domestication of Native Amazonian Crops. *Diversity*, 2(1), 72–106.

Coreau, A., Pinay, G., Thompson, J. D., Cheptou, P.-O., & Mermet, L. (2009). The rise of research on futures in ecology: rebalancing scenarios and predictions. *Ecology Letters*, *12*(12), 1277–1286.

Costanza, R., de Groot, R., Sutton, P., van der Ploeg, S., Anderson, S. J., Kubiszewski, I., Farber, S., & Turner, R. K. (2014). Changes in the global value of ecosystem services. *Global Environmental Change*, *26*, 152–158.

CRFM. (2014). *Sub-Regional Fisheries Management Plan for Flyingfish in the Eastern Caribbean. CRFM Special Publication No. 2*. Retrieved from http://www.fao.org/fi/static-media/MeetingDocuments/WECAFC16/Ref19e.pdf

Crosby, A. W. (1987). *The Columbian voyages, the Columbian exchange, and their historians*. Washington DC: American Historical Association. Retrieved from https://quod.lib.umich.edu/cgi/t/text/text-idx?c=acls;cc=acls;view=toc;idno=heb03124.0001.001;rgn=full text

Cultural Survival Quarterly Magazine (2013). Belize: our life, our lands-respect Maya land rights (2013). Retrieved from https://www.culturalsurvival.org/take-action/belize-our-life-our-lands-respect-maya-land-rights/belize-our-life-our-lands-respect

Cultural Survival Quarterly Magazine (2017). YURUMEIN (Our Homeland): A Film About Garifuna Cultural Renaissance on St. Vincent. Retrieved from https://www.culturalsurvival.org/news/yurumein-our-homeland-film-about-garifuna-cultural-renaissance-st-vincent

Das, P., & Horton, R. (2017). Pollution, health, and the planet: time for decisive action. *The Lancet*, Retrieved from http://www.thelancet.com/commissions/pollution-and-health.

De Groot, R. S., Wilson, M. A., & Boumans, R. M. (2002). A typology for the classification, description and valuation of ecosystem functions, goods and services. *Ecological Economics*, 41(3), 393–408.

DeClerck, F. A. J., Chazdon, R., Holl, K. D., Milder, J. C., Finegan, B., Martinez-Salinas, A., Imbach, P., Canet, L., & Ramos, Z. (2010). Biodiversity conservation in human-modified landscapes of Mesoamerica: Past, present and future. *Biological Conservation*, *143*, 2301–2313.

Diamond, J. M. (1997). *Guns, germs, and steel : the fates of human societies*. New York: W.W. Norton & Co. Retrieved from http://books.wwnorton.com/books/Guns-Germs-and-Steel/

Diaz, R. J., & Rosenberg, R. (2008). Spreading dead zones and consequences for marine ecosystems. *Science*, 321(5891), 926-929.

Díaz, S., Demissew, S., Joly, C., Lonsdale, W. M., & Larigauderie, A. (2015). A Rosetta Stone for Nature ' s Benefits to People. *Plos Biology*, 1–8.

Dicks, L. V., Viana, B., Bommarco, R., Brosi, B., Arizmendi, M. del C., Cunningham, S. A., Galetto, L., Hill, R., Lopes, A. V., Pires, C., Taki, H., & Potts, S. G. (2016). Ten policies for pollinators. *Science*, *354*(6315), 975–976.

Driscoll, C. T., Buonocore, J. J., Levy, J. I., Lambert, K. F., Burtraw, D., Reid, S. B., Fakhraei, H., & Schwartz, J. (2015). US power plant carbon standards and clean air and health co-benefits. *Nature Climate Change*, *5*(6), 535–540.

Dudley, N., Harrison, I. J., Kettunen, M., Madgwick, J., & Mauerhofer, V. (2016). Natural solutions for water management of the future: freshwater protected areas at the 6th World Parks Congress. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 26(S1), 121–132.

EIA. (2017). Energy Information Administration Electricity Generation Dataset. EIA-906/920/923 Data File. U.S. Energy Information Administration. Retrieved from https://www.eia.gov/electricity/data/eia923/

Engel, S., Pagiola, S., & Wunder, S. (2008). Designing payments for environmental services in theory and practice: An overview of the issues. *Ecological Economics*, 65(4), 663–674.

Estado Plurinacional de Bolivia (2015). Vivir Bien en armonía con la Madre Tierra. V Informe Nacional. Convenio de las Naciones Unidas sobre la Biodiversidad Biológica (CBD).

FAO. (2012a). Why the Andes Matter. Sustainable Mountain Development RIO 2012 and beyond. Food and Agriculture Organization of United Nations. Retrieved from http://www.fao.org/fileadmin/user_upload/mountain_partnership/docs/Print_E_Brief_Andes-low.pdf

FAO. (2012b). Forest tenure in Latin America. Retrieved from http://www.fao.org/forestry/54368/es/guy/.

FAO. (2013). The Outlook for Agriculture and Rural Development in the Americas. Food and Agriculture Organization of United Nations. Retrieved from http://www.fao.org/publications/card/en/c/daba72ad-6b98-5a13-88d8-83e796e6c4fa/

FAO. (2015). Fishers' knowledge and the ecosystem approach to fisheries. Regional Office for Latin America and the Caribbean. Food and Agriculture Organization of United Nations. Retrieved from http://www.fao.org/publications/card/en/c/9b643e50-fc20-4565-9962-bdf6d072dd32/

Feeley, T. J., Skone, T. J., Stiegel, G. J., McNemar, A., Nemeth, M., Schimmoller, B., Murphy, J. T., & Manfredo, L. (2008). Water: A critical resource in the thermoelectric power industry. *Energy*, *33*(1), 1–11. https://doi.org/10.1016/j.energy.2007.08.007

Ferrario, F., Beck, M. W., Storlazzi, C. D., Micheli, F., Shepard, C. C., & Airoldi, L. (2014). The effectiveness of coral reefs for coastal hazard risk reduction and adaptation. *Nature Communications*, 5, 4794.

Ferraro, P. J., Hanauer, M. M., Miteva, D. A., Nelson, J. L., Pattanayak, S. K., Nolte, C., & Sims, K. R. E. (2015). Estimating the impacts of conservation on ecosystem services and poverty by integrating modeling and evaluation. *PNAS*, 112(24), 7420–5.

Foley, J. a, Defries, R., Asner, G. P., Barford, C., Bonan, G., Carpenter, S. R., Chapin, F. S., Coe, M. T., Daily, G. C., Gibbs, H. K., Helkowski, J. H., Holloway, T., Howard, E. a, Kucharik, C. J., Monfreda, C., Patz, J. a, Prentice, I. C., Ramankutty, N., & Snyder, P. K. (2005). Global consequences of land use. *Science*, *309*(5734), 570–574.

Foley, J. A., Ramankutty, N., Brauman, K. A., Cassidy, E. S., Gerber, J. S., Johnston, M., Mueller, N. D., O'Connell, C., Ray, D. K., West, P. C., Balzer, C., Bennett, E. M., Carpenter, S. R., Hill, J., Monfreda, C., Polasky, S., Rockström, J., Sheehan, J., Siebert, S., Tilman, D., & Zaks, D. P. M. (2011). Solutions for a cultivated planet. *Nature, 478, 337*–342.

Franklin, J., Ripplinger, J., Freid, E. H., Marcano-Vega, H., & Steadman, D. W. (2015). Regional variation in Caribbean dry forest tree species composition. *Plant Ecology*, *216*(6), 873–886.

Fuller, R. A., Irvine, K. N., Devine-Wright, P., Warren, P. H., & Gaston, K. J. (2007). Psychological benefits of greenspace increase with biodiversity. *Biology Letters*, 3(4), 390–4.

Galluzzi, G., Eyzaguirre, P., & Negri, V. (2010). Home gardens: neglected hotspots of agro-biodiversity and cultural diversity. *Biodiversity and Conservation*, 19(13), 3635–3654.

Gander, M. J. (2014). International water law and supporting water management principles in the development of a model transboundary agreement between riparians in international river basins. *Water International*, 39(3), 315–332.

Garbach, K., Milder, J. C., DeClerck, F. A., Montenegro de Wit, M., Driscoll, L., & Gemmill-Herren, B. (2016). Examining multi-functionality for crop yield and ecosystem services in five systems of agroecological intensification. *International Journal of Agricultural Sustainability*, 15(1), 11-28.

Garbach, K., Milder, J., Montenegro, M., Karp, D., & DeClerck, F. (2014). Biodiversity and Ecosystem Services in Agroecosystems. *Encyclopedia of Agriculture and Food Sys*tems, 2, 21–40.

Garreaud, R. D., Vuille, M., Compagnucci, R., & Marengo, J. (2009). Present-day South American climate. *Palaeogeography, Palaeoclimatology, Palaeoecology*, *281*(3–4), 180–195.

Gibbs, H. K., Rausch, L., Munger, J., Schelly, I., Morton, D. C., Noojipady, P., Soares-Filho, B., Barreto, P., Micol, L., & Walker, N. F. (2015). Environment and development. Brazil's Soy Moratorium. *Science*, *347*(6220), 377–378.

Gleeson, T., Wada, Y., Bierkens, M. F. P., & van Beek, L. P. H. (2012). Water balance of global aquifers revealed by groundwater footprint. *Nature*, 488(7410), 197–200.

Global Footprint Network. (2016). National Footprint Accounts, 2016 Edition.

Gomez-Baggethun, E., Barton, D., Berry, P., Dunford, R., & Harrison, P. (2014). Concepts and Methods in Ecosystem Services Valuation. In M. Potschin, R. Haines-Young, R. Fish, & R. K. Turner (Eds.), *Routledge Handbook of Ecosystem Services* (pp. 99–111). London and New York: Routledge.

Granger, O. E. (1997). Caribbean Island States: Perils and Prospects in a Changing Global Environment. *Journal of Coastal Research*, (24), 71–93.

Green, P., C.J. Vörösmarty, I. Harrison, T. Farrell, L. Sáenz, B.M. Fekete. (2015). Freshwater ecosystem services supporting humans: Pivoting from water crisis to water solutions. *Global Environmental Change*, 34, 108-118.

Gregor Barié, C. (2014). Nuevas narrativas constitucionales en Bolivia y Ecuador: el buen vivir y los derechos de la naturaleza. Latinoamérica. *Revista de Estudios Latinoamericanos*, 59, 9–40.

Griggs, D., Stafford-Smith, M., Gaffney, O., Rockström, J., Öhman, M. C., Shyamsundar, P., Steffen, W., Glaser, G., Kanie, N., & Noble, I. (2013). Sustainable development goals for people and planet. *Nature*, *495*(7441), 305–307. https://doi.org/10.1038/495305a

Grizzetti, B., Lanzanova, D., Liquete, C., Reynaud, A., & Cardoso, A. C. (2016). Assessing water ecosystem services for water resource management. *Environmental Science & Policy*, 61, 194–203.

Guardiola, J., & García-Quero, F. (2014). Buen Vivir (living well) in Ecuador: Community and environmental satisfaction without household material prosperity? *Ecological Economics*, 107, 177–184.

Guedes, G. R., Brondízio, E. S., Barbieri, A. F., Anne, R., Penna-Firme, R., & D'Antona, Á. O. (2012). Poverty and Inequality in the Rural Brazilian Amazon: A Multidimensional Approach. *Human Ecology*, 40(1), 41–57.

Gygli, S., Haelg, F., & Sturm, J.-E. (2018). The KOF Globalisation Index – Revisited. *KOF Working Papers*. Zurich: KOF Swiss Economic Institute. Retrieved from https://www.ethz.ch/content/dam/ethz/special-interest/dual/kof-dam/documents/Globalization/2018/KOF_Globalisation_Index_Revisited.pdf

Haines-Young, R., & Potschin, M. (2012). Common international classification of ecosystem services (CICES, Version 4.1). European Environment Agency, 33.

Hansen, K.B., Jepsen, Käthe; Jacquelin, Pamela L. (2017). The Indigenous World Copenhagen: International Work Group for Indigenous Affairs (IWGIA). Retrieved from http://www.iwgia.org/publications/search-pubs?publication_id=760

Hanson, C., Talberth, J., & Yonavjak, L. (2011). Forests for water: Exploring Payments for Watershed Services in the U.S. South. Washington DC: World Resources Institute. Retrieved from http://pdf.wri.org/forests_for_water.pdf

Harrison, I. J., Green, P. A., Farrell, T. A., Juffe-Bignoli, D., Sáenz, L., & Vörösmarty, C. J. (2016). Protected areas and freshwater provisioning: a global assessment of freshwater provision, threats and management strategies to support human water security. *Aquatic Conservation: Marine and Freshwater Ecosystems*, *26*(S1), 103–120.

Hawley, K., Moench, M., & Sabbag, L. (2012). *Understanding the Economics of Flood Risk Reduction: A Preliminary Analysis*. Boulder: Institute for Social and Environmental Transition.

Hermoso, V., Abell, R., Linke, S., & Boon, P. (2016). The role of protected areas for freshwater biodiversity conservation: challenges and opportunities in a rapidly changing world. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 26(S1), 3–11.

Hughes, T. P. (1994). Catastrophes, phase shifts, and large-scale degradation of a Caribbean coral reef. *Science*, *265*(5178), 1547–51.

IBGE. (2017). Instituto Brasileiro de Geografia e Estatística. Países@. Retrieved from http://paises.ibge.gov.br/#/pt/pais

IEA (2015). Energy Climate and Change. World Energy Outlook Special Report. (International Energy Agency). Paris.

Instituto Nacional de Estadística y Geografia México (2015). La Encuesta Intercensal 2015. Retrieved from http://www.beta.inegi.org.mx/proyectos/enchogares/especiales/intercensal/

Instituto Socioambiental (ISA). (2010). Populações indígenas no Brasil. Retrieved from https://pib.socioambiental.org/pt/c/0/1/2/populacao-indigena-no-brasil

Ioris, A. A. R. (2016). The paradox of poverty in rich ecosystems: impoverishment and development in the Amazon of Brazil and Bolivia. *The Geographical Journal*, 182(2), 178–189.

IPBES (2016a). Summary for policymakers of the assessment report of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services on pollinators, pollination and food production. S.G. Potts, V. L. Imperatriz-Fonseca, H. T. Ngo, J. C. Biesmeijer, T. D. Breeze, L. V. Dicks, L. A. Garibaldi, R. Hill, J. Settele, A. J. Vanbergen, M. A. Aizen, S. A. Cunningham, C. Eardley, B. M. Freitas, N. Gallai, P. G. Kevan, A. Kovács-Hostyánszki, P. K. Kwapong, J. Li, X. Li, D. J. Martins, G. Nates-Parra, J. S. Pettis, R. Rader, and B. F. Viana (eds.). Secretariat of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services, Bonn, Germany. 36 pages. Retieved from www.ipbes.net/sites/default/files/downloads/pdf/spm_deliverable_3a_pollination_20170222.pdf

IPBES. (2014). Guide on the production and integration of assessments from and across all scales (deliverable 2 (a)). Bonn: IPBES/3/INF/4. Secretariat of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services.

IPBES. (2015a). Report of the Plenary of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services on the work of its third session. Annex V Scoping for a regional assessment of biodiversity and ecosystem services and functions for the Americas. Bonn: Secretariat of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services.

IPBES. (2015b). Preliminary guide regarding diverse conceptualization of multiple values of nature and its benefits, including biodiversity and ecosystem functions and services (deliverable 3 (d)). Kuala Lumpur: IPBES/4/INF/13. Plenary of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services Fourth session.

IPBES. (2016b). The methodological assessment report on Scenarios and Models of Biodiversity and Ecosystem Services of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. In S. Ferrier, K. N. Ninan, P. Leadley, R. Alkemade, L. A. Acosta, H. R. Akçakaya, L. Brotons, W. W. L. Cheung, V. Christensen, K. A. Harhash, J. Kabubo-Mariara, C. Lundquist, M. Obersteiner, H. M. Pereira, G. Peterson, R. Pichs-Madruga, N. Ravindranath, C. Rondinini, & B. A. Wintle (Eds.) (p. 348). Bonn, Germany: Secretariat of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. Retrieved from https://www.ipbes.net/sites/default/files/downloads/pdf/2016.methodological_assessment_report_scenarios_models.pdf

IPBES. (2017a). Update on the classification of nature's contributions to people by the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services, Germany: IPBES/5/INF/24. Plenary of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services Fifth session. Available from: https://www.ipbes.net/system/tdf/downloads/pdf/ipbes-5-inf-24.pdf?file=1&type=node&id=534

IPBES. (2017b). Update on the work on knowledge and data (deliverables 1 (d) and 4 (b)). Germany: IPBES/5/INF/5. Plenary of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services Fifth session. Available from: https://www.ipbes.net/system/tdf/downloads/pdf/ipbes-5-inf-5.pdf?file=1&type=node&id=540

Jacobsen, S.-E., Sørensen, M., Pedersen, S. M., & Weiner, J. (2013). Feeding the world: genetically modified crops versus agricultural biodiversity. *Agronomy for Sustainable Development*, *33*(4), 651–662.

Jenkins, C. N., Pimm, S. L., & Joppa, L. N. (2013). Global patterns of terrestrial vertebrate diversity and conservation. *PNAS*, *110*(28), E2602-10.

Jetoo, S., Thorn, A., Friedman, K., Gosman, S., & Krantzberg, G. (2015). Governance and geopolitics as drivers of change in the Great Lakes-St. Lawrence basin. *Journal of Great Lakes Research*, *41*(S1), 108–118.

Johns, C. M. (2017). The Great Lakes, Water Quality and Water Policy in Canada. In S. Renzetti & D. P. Dupont (Eds.) (pp. 159–178). Springer.

Joppa, L. N., & Pfaff, A. (2009). High and Far: Biases in the Location of Protected Areas. *PloS ONE*, 4(12), e8273.

Juffe-Bignoli, D., Burgess, N.D., Bingham, H., Belle, E.M.S., de Lima, M.G., Deguignet, M., Bertzky, B., Milam, A.N., Martinez-Lopez, J., Lewis, E., Eassom, A., Wicander, S., Geldmann, J., van Soesbergen, A., Arnell, A.P., O'Connor, B., Park, S., Shi, Y.N., Danks, F.S., MacSharry, B., Kingston, N. (2014). *Protected Planet Report 2014*. Cambridge, UK: UNEP-WCMC. Retrieved from http://wdpa.s3.amazonaws.com/WPC2014/protected_planet_report.pdf.

Juffe-Bignoli, D., Harrison, I., Butchart, S. H., Flitcroft, R., Hermoso, V., Jonas, H., Lukasiewicz, A., Thieme, M., Turak, E., Bingham, H., Dalton, J., Darwall, W., Deguignet, M., Dudley, N., Gardner, R., Higgins, J., Kumar, R., Linke, S., Milton, G. R., Pittock, J., Smith, K. G., & van Soesbergen, A. (2016). Achieving Aichi Biodiversity Target 11 to improve the performance of protected areas and conserve freshwater biodiversity. *Aquatic Conservation: Marine and Freshwater Ecosystems*, *26*, 133–151. https://doi.org/10.1002/aqc.2638

Kalinago Territory (2017). Home to the indigenous people of Dominica. Dominica's First Early Inhabitants. Retrieved from http://kalinagoterritory.com/about-us/

Kaplan, H., Randall, C., Thompson, M. D., Benjamin, C., Trumble, & Al., E. (2017). Coronary atherosclerosis in indigenous South American Tsimane: a cross-sectional cohort study. *The Lancet*, 389(10080), 1730–1739.

Kaufmann, D., Kraay, A., & Mastruzzi, M. (2010, September 1). The Worldwide Governance Indicators: Methodology and Analytical Issues. World Bank Policy Research Woking Paper No. 5430.

Kimbell, A. R., & Brown, H. (2009). Using Forestry to Secure America's Water Supply. *Journal of Forestry*, 107(3), 146–149.

Kipuri, N. (2009). Culture. In State of the World's Indigenous Peoples. United Nations Department of Economic and Social Affairs, New York.

Klein, A. M., Müller, C., Hoehn, P., & Kremen, C. (2009). Understanding the role of species richness for crop pollination services. In S. Naeem, D. Bunker, A. Hector, M. Loreau, & C. Perrings (Eds.), *Biodiversity, Ecosystem Functioning, and Human Wellbeing: An Ecological and Economic Perspective* (pp. 195–208). Oxford University Press.

Knowlton, K., Rotkin-Ellman, M., Geballe L., Max, W., and G. M. Solomon. (2011). Six Climate Change–Related Events In The United States Accounted For About $14 Billion In Lost Lives And Health Costs. *Health Affairs*, (30)11, 2167-2176.

Kremen, C., & Miles, A. (2012). Ecosystem services in biologically diversified versus conventional farming systems: Benefits, externalities, and trade-offs. *Ecology and Society*, *17*(4), 40.

Kremen, C., & Ostfeld, R. S. (2005). A call to ecologists: measuring, analysing, and managing ecosystem services. *Frontiers in Ecology and the Environment*, *3*(10), 540–548.

Laurance, W. F. (1999). Reflections on the tropical deforestation crisis. *Biological Conservation*, *91*(2–3), 109–117.

Leb, C. (2015). One step at a time: international law and the duty to cooperate in the management of shared water resources. *Water International*, 40(1), 21–32.

Lehner, B., Verdin, K., Jarvis, A. (2006). HydroSHEDS Technical Documentation. World Wildlife Fund, Washington DC. Retrieved from http://hydrosheds.cr.usgs.gov.

Lopez, L. H. (2009). Reaching the unreached: indigenous intercultural bilingual education in Latin America. Paper commissioned for the EFA Global Monitoring Report 2010. UNESCO. Retrieved from http://unesdoc.unesco.org/images/0018/001866/186620e.pdf

Lugo, A. E. (1995). Management of Tropical Biodiversity. *Ecological Applications*, *5*(4), 956–961.

Lugo, A. E., Schmidt, R., & Brown, S. (1981). Tropical forests in the Caribbean. *Ambio*, *10*(6), 318–324.

Macknick, J., Newmark, R., Heath, G., & Hallett, K. (2011). A Review of Operational Water Consumption and Withdrawal Factors for Electricity Generating Technologies. Colorado. Retrieved from https://www.nrel.gov/docs/fy11osti/50900.pdf

Maller, C., Townsend, M., Pryor, A., Brown, P., & St Leger, L. (2006). Healthy nature healthy people: 'contact with nature' as an upstream health promotion intervention for populations. *Health Promotion International*, *21*(1), 45-54.

Marchant, R. (2010). Understanding complexity in savannas: climate, biodiversity and people. *Current Opinion in Environmental Sustainability*, *2*(1–2), 101–108.

Marchese, C. (2015). Biodiversity hotspots: A shortcut for a more complicated concept. *Global Ecology and Conservation*, 3, 297–309.

Marengo, J. A. (2006). On the hydrological cycle of the Amazon Basin: A Historical Review and Current State-of-the-Art. *Revista Brasileira de Metereologia*, 21(3), 1–19.

Martin, A., Coolsaet, B., Corbera, E., Dawson, N. M., Fraser, J. A., Lehmann, I., & Rodriguez, I. (2016). Justice and conservation: The need to incorporate recognition. *Biological Conservation*, *197*, 254–261.

Matson, P. A., Parton, W. J., Power, A. G., & Swift, M. J. (1997). Agricultural intensification and ecosystem properties. *Science*, 277(5325), 504-509.

McDonald, R. I., Weber, K. F., Padowski, J., Boucher, T., & Shemie, D. (2016). Estimating watershed degradation over the last century and its impact on water-treatment costs for the world's large cities. *PNAS*, 113(32), 9117–22.

MEA. (2005). Millennium Ecosystem Assessment. Ecosystems and Human Well-being: Synthesis. Washington DC: Island Press.

Mekonnen M.M. and A.Y. Hoekstra. (2015). Global Gray Water Footprint and Water Pollution Levels Related to Anthropogenic Nitrogen Loads to Fresh Water. *Environmental Science and Technology*, 49 (21), 12860–12868

Mekonnen, M. M., & Hoekstra, A. Y. (2016). Four billion people facing severe water scarcity. *Science Advances*, 2(2), e1500323–e1500323.

Menton, M. C. S., Merry, F. D., Lawrence, A., & Brown, N. (2009). Company–Community Logging Contracts in Amazonian Settlements: Impacts on Livelihoods and NTFP Harvests. *Ecology and Society*, 14(1), 39.

Miloslavich, P., Klein, E., Díaz, J. M., Hernández, C. E., Bigatti, G., Campos, L., Artigas, F., Castillo, J., Penchaszadeh, P. E., Neill, P. E., Carranza, A., Retana, M. V., Díaz de Astarloa, J. M., Lewis, M., Yorio, P., Piriz, M. L., Rodríguez, D., Yoneshigue-Valentin, Y., Gamboa, L., & Martín, A. (2011). Marine Biodiversity in the Atlantic and Pacific Coasts of South America: Knowledge and Gaps. *PLoS ONE*, *6*(1), e14631. https://doi.org/10.1371/journal.pone.0014631

Ministerio de Desarrollo Social de Chile (2013). Pueblos Indigenas. Retrieved from http://observatorio.ministeriodesarrollosocial.gob.cl/documentos/Casen2013_Pueblos_Indigenas_13mar15_publicacion.pdf.

Miteva, D. A., Pattanayak, S. K., & Ferraro, P. J. (2012). Evaluation of biodiversity policy instruments: what works and what doesn't? *Oxford Review of Economic Policy*, *28*(1), 69–92.

Montenegro, R. A., & Stephens, C. (2006). Indigenous health in Latin America and the Caribbean. *The Lancet*, *367*(9525), 1859–1869.

Morello-Frosch, R., Zuk, M., Jerrett, M., Shamasunder, B., & Kyle, A. D. (2011). Understanding The Cumulative Impacts Of Inequalities In Environmental Health: Implications For Policy. *Health Affairs*, *30*(5), 879–887.

Mueller, J. M., Swaffar, W., Nielsen, E. A., Springer, A. E., & Lopez, S. M. (2013). Estimating the value of watershed services following forest restoration. *Water Resources Research*, 49(4), 1773–1781.

Myers, N., Mittermeier, R. A., Mittermeier, C. G., da Fonseca, G. A. B., & Kent, J. (2000). Biodiversity hotspots for conservation priorities. *Nature*, 403(6772), 853–858.

Nazareno, A. G., & Laurance, W. F. (2015). Brazil's drought: Beware deforestation. *Science*, 347(6229), 1427.

Nepstad, D. C., Stickler, C. M., Filho, B. S.-, & Merry, F. (2008). Interactions among Amazon land use, forests and climate: prospects for a near-term forest tipping point. *Philosophical Transactions of the Royal Society of London. Series B, Biological Sciences*, *363*(1498), 1737–46.

Nepstad, D., McGrath, D., Stickler, C., Alencar, A., Azevedo, A., Swette, B., Bezerra, T., DiGiano, M., Shimada, J., Seroa da Motta, R., Armijo, E., Castello, L., Brando, P., Hansen, M. C., McGrath-Horn, M., Carvalho, O., & Hess, L. (2014). Slowing Amazon deforestation through public policy and interventions in beef and soy supply chains. *Science*, *344*(6188), 1118–1123. https://doi.org/10.1126/science.1248525

Nilsson, C., Reidy, C. A., Dynesius, M., & Revenga, C. (2005). Fragmentation and flow regulation of the world's large river systems. *Science*, *308*(5720), 405–8.

Nugent, D., & Sovacool, B. K. (2014). Assessing the lifecycle greenhouse gas emissions from solar PV and wind energy: A critical meta-survey. *Energy Policy*, *65*, 229–244.

Oliveira, P. J. C., Asner, G. P., Knapp, D. E., Almeyda, A., Galvan-Gildemeister, R., Keene, S., Raybin, R. F., & Smith, R. C. (2007). Land-Use Allocation Protects the Peruvian Amazon. *Science*, *317*(5842), 1233–1236. https://doi.org/10.1126/science.1146324

Olson, D. M., & Dinerstein, E. (2002). The Global 200: Priority Ecoregions for Global Conservation. *Annals of the Missouri Botanical Garden*, *89*(2), 199.

Olson, D. M., Dinerstein, E., Wikramanayake, E. D., Burgess, N. D., Powell, G. V. N., Underwood, E. C., D'amico, J. a., Itoua, I., Strand, H. E., Morrison, J. C., Loucks, C. J., Allnutt, T. F., Ricketts, T. H., Kura, Y., Lamoreux, J. F., Wettengel, W. W., Hedao, P., & Kassem, K. R. (2001). Terrestrial Ecoregions of the World: A New Map of Life on Earth. *BioScience*, *51*(11), 933. https://doi.org/10.1641/0006-3568(2001)051[0933:TEOTWA]2.0.CO;2

Pabon-Zamora, L., A. Fauzi, A. Halim, J. Bezaury-Creel, E. Vega-Lopez, F. Leon, L. Gil & V. Cartaya (2008): Protected areas and human well-being: Experiences from Indonesia, Mexico, Peru and Venezuela. CBD Technical Series No. 36, Secretariat of Convention on Biological Diversity, Montreal, Canada

Pacheco, D. (2014). Hacia la descolonización de las políticcas ambientales y de los bosques: el mecanismo conjunto de mitigación y adaptación para el manejo integral y sustentable de los bosques y la Madre Tierra. La Paz: Fundación de la Cordillera, Universidad de la Cordillera.

Parra, F., & Casas, A. (2016). Origen y difusión de la domesticación y la agricultura en el Nuevo Mundo. In A. Casas, J. T.-G. Y, & F. Parra (Eds.), *Domesticación en el Continente Americano. Volumen 1. Manejo de biodiveridad y evolución dirigida por las culturas del Nuevo Mundo* (pp. 159–184). Universidad Nacional Autónoma de México /Universidad Nacional Agraria La Molina.

Pascual, U., Balvanera, P., Díaz, S., Pataki, G., Roth, E., Stenseke, M., Watson, R. T., Başak Dessane, E., Islar, M., Kelemen, E., Maris, V., Quaas, M., Subramanian, S. M., Wittmer, H., Adlan, A., Ahn, S., Al-Hafedh, Y. S., Amankwah, E., Asah, S. T., Berry, P., Bilgin, A., Breslow, S. J., Bullock, C., Cáceres, D., Daly-Hassen, H., Figueroa, E., Golden, C. D., Gómez-Baggethun, E., González-Jiménez, D., Houdet, J., Keune, H., Kumar, R., Ma, K., May, P. H., Mead, A., O'Farrell, P., Pandit, R., Pengue, W., Pichis-Madruga, R., Popa, F., Preston, S., Pacheco-Balanza, D., Saarikoski, H., Strassburg, B. B., van den Belt, M., Verma, M., Wickson, F., & Yagi, N. (2017). Valuing nature's contributions to people: the IPBES approach. *Current Opinion in Environmental Sustainability*, *26–27*, 7–16. https://doi.org/10.1016/J.COSUST.2016.12.006

Pascual, U., Phelps, J., Garmendia, E., Brown, K., Corbera, E., Martin, A., Gomez-Baggethun, E., & Muradian, R. (2014). Social Equity Matters in Payments for Ecosystem Services. *BioScience*, *64*(11), 1027–1036. https://doi.org/10.1093/biosci/biu146

Perrings, C., Duraiappah, A., Larigauderie, A., & Mooney, H. (2011). The Biodiversity and Ecosystem Services Science-Policy Interface. *Science*, *331*(6021).

Perry, C. T., Murphy, G. N., Kench, P. S., Smithers, S. G., Edinger, E. N., Steneck, R. S., & Mumby, P. J. (2013). Caribbean-wide decline in carbonate production threatens coral reef growth. *Nature Communications*, *4*, 1402.

Perz, S., Leite, F., Griffin, L., Hoelle, J., Rosero, M., Carvalho, L., Castillo, J., & Rojas, D. (2015). Trans-Boundary Infrastructure and Changes in Rural Livelihood Diversity in the Southwestern Amazon: Resilience and Inequality. *Sustainability*, *7*(9), 12807–12836. https://doi.org/10.3390/su70912807

Phillips, O. L., & Brienen, R. J. W. (2017). Carbon uptake by mature Amazon forests has mitigated Amazon nations' carbon emissions. *Carbon Balance and Management*, 12(1), 1–9.

Pilgrim, S., & Pretty, J. N. (2010). *Nature and Culture: Rebuilding Lost Connections*. London, Washington DC.: Earthscan.

Pinho, P. F., Patenaude, G., Ometto, J. P., Meir, P., Toledo, P. M., Coelho, A., & Young, C. E. F. (2014). Ecosystem protection and poverty alleviation in the tropics: Perspective from a historical evolution of policy-making in the Brazilian Amazon. *Ecosystem Services*, 8, 97–109.

Pires, G. F., & Costa, M. H. (2013). Deforestation causes different subregional effects on the Amazon bioclimatic equilibrium. *Geophysical Research Letters*, 40(14), 3618–3623.

PNUD (2013). Atlas Desenvolvimento Humano no Brazil 2013. Programa das Nações Unidas para o Desenvolvimento/ Instituto de Pesquisa Econômica Aplicada/ Fundação João Pinheiro. Retrieved from http://atlasbrasil.org.br/2013/

Poole, R. (2011). What became of the Taíno. Smithsonian Magazine. Retrieved from http://www.smithsonianmag.com/people-places/what-became-of-the-taino-73824867/#2zoQt67hghZcgmpZ.99

Postel, S. L. (2000). Entering an Era of Water Scarcity: The Challenges Ahead. *Ecological Applications*, 10(4), 941–948.

Ramsar. (2008). The Changwon Declaration on human well-being and wetlands. Retrieved from http://ramsar.rgis.ch/pdf/cop10/cop10_changwon_english.pdf

Raudsepp-Hearne, C., Peterson, G. D., Tengö, M., Bennett, E. M., Holland, T., Benessaiah, K., Macdonald, G. K., & Pfeifer, L. (2010). Untangling the Environmentalist's Paradox: Why is Human Well-Being Increasing as Ecosystem Services Degrade? *Source: BioScience Articles BioScience*, *60*(8), 576–589. https://doi.org/10.1525/bio.2010.60.8.4

Ribeiro, M. C., Metzger, J. P., Camargo Martensen, A., Ponzoni, F. J., & Hirota, M. M. (2009). The Brazilian Atlantic Forest: How much is left, and how is the remaining forest distributed? Implications for conservation. *Biological Conservation*, *142*, 1141–1153.

Rice, J., Gjerde, K. M., Ardron, J., Arico, S., Cresswell, I., Escobar, E., Grant, S., & Vierros, M. (2011). Policy relevance of biogeographic classification for conservation and management of marine biodiversity beyond national jurisdiction, and the GOODS biogeographic classification. *Ocean and Coastal Management*, *54*, 110–122. https://doi.org/10.1016/j.ocecoaman.2010.10.010

Roberts, C. M., McClean, C. J., Veron, J. E. N., Hawkins, J. P., Allen, G. R., McAllister, D. E., Mittermeier, C. G., Schueler, F. W., Spalding, M., Wells, F., Vynne, C., & Werner, T. B. (2002). Marine Biodiversity Hotspots and Conservation Priorities for Tropical Reefs. *Science*, *295*(5558), 1280–1284. https://doi.org/10.1126/science.1067728

Rodríguez Osuna, V., Navarro Sánchez, G., Sommer, J. H., & Biber-Freudenberger, L. (2017). Towards the Integration of Biodiversity in Environmental Impact Assessments of Bolivia. Cochabamba: Editoria INIA. Center for Development Research (ZEF)-Universidad Católica Boliviana (UCB).

Rodríguez Osuna, V., P. May, J. Monteiro, R. Wollenweber, H. Hissa, & Costa, M. (2018). Promoting Sustainable Agriculture, Boosting Productivity and Enhancing Climate Mitigation and Adaptation Through the Rio Rural Program, Brazil. Chapter 28. In: Nehren, U., S. Schlueter, C. Raedig, D. Sattler, and H. Hissa (eds.), Strategies and tools for a sustainable Rio de Janeiro. Springer International Publishing.

Russi, D., Ten Brink, P., Farmer, A., & Badura, T. (2013). TEEB for Water and Wetlands. London, Brussels. Retrieved from http://www.ramsar.org/sites/default/files/documents/library/teeb_waterwetlands_report_2013.pdf

Santa Rosa First Peoples Community (2015). The establishment of an amerindian heritage village and living museum for the Santa Rosa First Peoples' community of Arima at Blanchisseuse Road, Arima: excerpts of a preliminary master plan. Retrieved from http://santarosafirstpeoples.org/resource-center/an-amerindian-heritage-complex/

Scarano, F. R., & Ceotto, P. (2015). Brazilian Atlantic forest: impact, vulnerability, and adaptation to climate change. Biodiversity and Conservation, 24, 2319–2331.

Schaaf T. & Lee, C. (2006). Conserving cultural and biological diversity: The role of sacred natural sites and cultural landscapes. International symposium held in Tokyo, Japan, 30 May-2June 2005. Paris: United Nations

Schmeller, D. S., & Bridgewater, P. (2016). The Intergovernmental Platform on Biodiversity and Ecosystem Services (IPBES): progress and next steps. *Biodiversity and Conservation*, *25*(5), 801–805.

Schwanitz, V. J., Piontek, F., Bertram, C., & Luderer, G. (2014). Long-term climate policy implications of phasing out fossil fuel subsidies. *Energy Policy*, *67*, 882–894.

Siegel, K. M. (2017). Regional Environmental Cooperation in the La Plata River Basin. In Regional Environmental Cooperation in South America (pp. 91–121). London: Palgrave Macmillan UK.

Spalding, M. D., Fox, H. E., Allen, G. R., Davidson, N., Ferdaña, Z. A., Finlayson, M., Halpern, B. S., Jorge, M. A., Lombana, A., Lourie, S. A., Martin, K. D., Mcmanus, E., Molnar, J., Recchia, C. A., & Robertson, J. (2007). Marine Ecoregions of the World: A Bioregionalization of Coastal and Shelf Areas. *Source: BioScience*, *57*(7), 573–583. https://doi.org/10.1641/B570707

Statistics Canada (2017). Aboriginal peoples Retrieved from http://www.statcan.gc.ca/eng/start

TEEB. (2009). TEEB for National and International Policy Makers. Summary: Responding to the Value of Nature 2009. the Approach, Conclusions and Recommendations of TEEB. Wesseling

TEEB. (2010). Mainstreaming the Economics of Nature: A Synthesis of the Approach, Conclusions and Recommendations of TEEB. The Economics of Ecosystems and Biodiversity, Malta

Tengö, M., Brondizio, E. S., Elmqvist, T., Malmer, P., & Spierenburg, M. (2014). Connecting Diverse Knowledge Systems for Enhanced Ecosystem Governance: The Multiple Evidence Base Approach. *AMBIO*, 43(5), 579–591.

The Economist (2017). Pocket World in Figures. The Economist Newspaper Ltd. Profile Books Ltd. United States

Tscharntke, T., Klein, A. M., Kruess, A., Steffan-Dewenter, I., & Thies, C. (2005). Landscape perspectives on agricultural intensification and biodiversity–ecosystem service management. *Ecology letters*, 8(8), 857-874.

Tundisi, J. G., Goldemberg, J., Matsumura-Tundisi, T., & Saraiva, A. C. F. (2014). How many more dams in the Amazon? *Energy Policy*, 74, 703–708.

UN (2016a). Summary of the First Global Integrated Marine Assessment. Retrieved from http://www.un.org/Depts/los/global_reporting/WOA_RPROC/Summary.pdf

UN. (2013). World Population Prospects. The 2012 Revision. Highlights and Advance Tables. United Nations New York. Retrieved from https://esa.un.org/unpd/wpp/publications/Files/WPP2012_HIGHLIGHTS.pdf

UN. (2015). Transforming our world: the 2030 Agenda for Sustainable Development. Paris: UN General Assembly. Retrieved from https://sustainabledevelopment.un.org/post2015/transformingourworld

UN. (2016b). Paris Agreement. Paris: United Nations Framework Convention on Climate Change. Retrieved from http://unfccc.int/files/essential_background/convention/application/pdf/english_paris_agreement.pdf

UN-DESA. (2014). World Urbanization Prospects*: The 2014 Revision, Highlights (ST/ESA/SER.A/352).* United Nations, Department of Economic and Social Affairs, Population Division. Retrieved from https://esa.un.org/unpd/wup/publications/files/wup2014-highlights.pdf

UN-DESA. (2016). The World's Cities in 2016 (Statistical Papers - United Nations (Ser. A), Population and Vital Statistics Report). New York: United Nations. https://doi.org/10.18356/8519891f-en

UNDP. (2016). Human Development Report 2016. Human Development for Everyone. United Nations Development Programme. Retrieved from http://hdr.undp.org/en/2016-report

UNEP. (2012). Global Environment Outlook. GEO-5: Environment for the future we want. United Nations Environment Programme, Malta. Retrieved from http://web.unep.org/geo/assessments/global-assessments/global-environment-outlook-5

UNEP-DHI and UNEP (2016). Transboundary River Basins: Status and Trends. Volume 3: River Basins. United Nations Environment Programme, Nairobi. Retrieved from http://www.geftwap.org/publications/river-basins-spm

UNESCO. (2017). Concept of Governance. United Nations Educational, Scientific and Cultural Organization. Retrieved from http://www.unesco.org/new/en/education/themes/strengthening-education-systems/quality-framework/technical-notes/concept-of-governance

US Department of the Interior Indians Affair (2017). Retrieved from https://www.bia.gov/FAQs/index.htm.

Van Dam, C. (2011). Indigenous territories and REDD in Latin America: Opportunity or threat? *Forests*, 2(1), 394-414.

Van Zanten, B. T., Van Beukering, P. J. H., & Wagtendonk, A. J. (2014). Coastal protection by coral reefs: A framework for spatial assessment and economic valuation. *Ocean and Coastal Management*, 96, 94–103.

Veiga, J. B., Tourrand, J. F., Piketty, M. G., Poccard-Chapuis, R., Alves, A. M., & Thales, M. C. (2004). Expansão e Trajetórias da Pecuária na Amazônica: Pará, Brasil. Brasillia: Editora Universidade de Brasilia.

Venter, O., Sanderson, E. W., Magrach, A., Allan, J. R., Beher, J., Jones, K. R., Possingham, H. P., Laurance, W. F., Wood, P., Fekete, B. M., Levy, M. A., & Watson, J. E. M. (2016). Sixteen years of change in the global terrestrial human footprint and implications for biodiversity conservation. *Nature Communications*, *7*, 12558. https://doi.org/10.1038/ncomms12558

Vogt, N., Pinedo-Vasquez, M., Brondízio, E. S., Rabelo, F. G., Fernandes, K., Almeida, O., Riveiro, S., Deadman, P. J., & Dou, Y. (2016). Local ecological knowledge and incremental adaptation to changing flood patterns in the Amazon delta. *Sustainability Science*, *11*(4), 611–623. https://doi.org/10.1007/s11625-015-0352-2

Vörösmarty, C. J., Osuna, V. R., Koehler, D. A., Klop, P., Spengler, J. D., Buonocore, J. J., Cak, A. D., Tessler, Z. D., Corsi, F., Green, P. A., & Sánchez, R. (2018). Scientifically assess impacts of sustainable investments. *Science (New York, N.Y.)*, *359*(6375), 523–525. https://doi.org/10.1126/science.aao3895

Watson, J. E. M., Dudley, N., Segan, D. B., & Hockings, M. (2014). The performance and potential of protected areas. *Nature*, 515(7525), 67–73.

WB &WWF (2003). The importance of forest protected areas to drinking water. Running Pure. Alliance for Forest Conservation and Sustainable Use. Gland/ Washington DC.: The World Bank/WWF Alliance. Retrieved from http://siteresources.worldbank.org/INTBIODIVERSITY/Resources/RunningPure2003+.pdf

WB (World Bank) & IHME (Institute for Health Metrics and Evaluation) (2016). The Cost of Air Pollution: Strengthening the Economic Case for Action. Washington DC: World Bank. Retrieved from https://openknowledge.worldbank.org/bitstream/handle/10986/25013/108141.pdf?sequence=4&isAllowed=y

WDPA. (2017). World Database on Protected Areas. ProtectedPlanet. Retrieved from https://www.protectedplanet.net/c/world-database-on-protected-areas

Winemiller, K. O., McIntyre, P. B., Castello, L., Fluet-Chouinard, E., Giarrizzo, T., Nam, S., Baird, I. G., Darwall, W., Lujan, N. K., Harrison, I., Stiassny, M. L. J., Silvano, R. A. M., Fitzgerald, D. B., Pelicice, F. M., Agostinho, A. A., Gomes, L. C., Albert, J. S., Baran, E., Petrere, M., Zarfl, C., Mulligan, M., Sullivan, J. P., Arantes, C. C., Sousa, L. M., Koning, A. A., Hoeinghaus, D. J., Sabaj, M., Lundberg, J. G., Armbruster, J., Thieme, M. L., Petry, P., Zuanon, J., Torrente Vilara, G., Snoeks, J., Ou, C., Rainboth, W., Pavanelli, C. S., Akama, A., van Soesbergen, A., & Sáenz, L. (2016). DEVELOPMENT AND ENVIRONMENT. Balancing hydropower and biodiversity in the Amazon, Congo, and Mekong. *Science (New York, N.Y.)*, *351*(6269), 128–129. https://doi.org/10.1126/science.aac7082

WOA. (2016). A Regular Process for Global Reporting and Assessment of the State of the Marine Environment, including Socio-economic Aspects. World Ocean Assessment. Division for Ocean Affairs and the Law of the Sea http://www.un.org/Depts/los/global_reporting/WOA_RegProcess.htm

Wood, S. L. R., & DeClerck, F. A. J. (2015). Ecosystems and human well-being in the Sustainable Development Goals. *Frontiers in Ecology and the Environment*, 13(3), 123.

World Atlas (2017). The longest coral reefs in the world. Retrieved from http://www.worldatlas.com/articles/the-longest-coral-reefs-in-the-world.html

World Bank. (2015). World Bank Open Data 2015. Retrieved from http://data.worldbank.org/indicator/SP.POP.TOTL

World Bank. (2017). Data retrieved from World Development Indicators Online (WDI) database.

World Database of Key Biodiversity Areas. (n.d.). Retrieved from www.keybiodiversityareas.org

Wuenscher, T., Engel, S., & Wunder, S. (2008). Spatial targeting of payments for environmental services: A tool for boosting conservation benefits. *Ecological Economics*, 65(4), 822–833.

WWAP. (2015). The United Nations World Water Development Report 2015: Water for a Sustainable World. Paris: United Nations World Water Assessment Programme. Retrieved from http://www.unesco.org/new/en/natural-sciences/environment/water/wwap/wwdr/2015-water-for-a-sustainable-world/

WWF. (2017). Mesoamerican Reef. Washington DC: World Wildlife Fund. Retrieved from https://www.worldwildlife.org/places/mesoamerican-reef#

Young, C. E. F., Aguiar, C., & Neto de Souza, E. (2015). Valorando Tempestades: Custo econômico dos eventos climáticos extremos no Brasil nos anos de 2002 – 2012. São Paulo: Observatório do Clima.

Zhang, D. Q., Jinadasa, K. B. S. N., Gersberg, R. M., Liu, Y., Ng, W. J., & Tan, S. K. (2014). Application of constructed wetlands for wastewater treatment in developing countries – A review of recent developments (2000–2013). *Journal of Environmental Management*, *141*, 116–131.

Chapter 2: Nature's contributions to people and quality of life

Coordinating Lead Authors:
Cristiana Simão Seixas (Brazil), Christopher B. Anderson (Argentina/USA), Siobhan Fennessy (USA), Bernal Herrera-F (Costa Rica).

Lead Authors:
Olga Barbosa (Chile), Richard Cole (USA), Rahanna Juman (Trinidad and Tobago), Laura Lopez-Hoffman (USA), Mónica Moraes R. (Bolivia), Gerhard Overbeck (Brazil/Germany), Wendy R. Townsend (Bolivia/US).

Fellow:
Julio Díaz-José (Mexico).

Contributing Authors:
Edgar Espinoza-Cisneros (Costa Rica), Marcello Hernández-Blanco (Costa Rica), Jake Rice (Canada), Simone Aparecida Veira (Brazil), Carlos Zembrana-Torrelio (Bolivia).

Review Editors:
Laura Nahuelhual (Chile), Brenda Parlee (Canada).

Suggested Citation:
Seixas, C. S., Anderson, C. B., Fennessy, S., Herrera-F., Bernal, Barbosa, O., Cole, R., Juman, R., Lopez-Hoffman, L., Moraes, R., M., Overbeck, G., Townsend, W. R., and Díaz-José, J., Chapter 2: Nature's contributions to people and quality of life. In IPBES (2018): The IPBES regional assessment report on biodiversity and ecosystem services for the Americas. Rice, J., Seixas, C. S., Zaccagnini, M. E., Bedoya-Gaitán, M., and Valderrama, N. (eds.). Secretariat of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services, Bonn, Germany, pp. xx-xx

Table of contents Chapter 2

2 Executive summary

1. **In the Americas, nature has an exceptional ability to contribute to human quality of life, due to its high biological diversity and productivity** (*well established*). Producing 40.5 per cent of the world's biocapacity, its residents have three times more resources per capita than an average global citizen {2.6, Table 2.24}, but availability and nature's benefits are not shared equitably among social groups, countries, and subregions {2.5, 2.6, 2.7, Figure 2.36} (*well established*).

2. **In the Americas, nature is used more intensively than the global average {2.6}**. The region hosts 13 per cent of the planet's human population, causing 22.8 per cent of the global ecological footprint; North America accounts for 63 per cent of the America's total {Table 2.24}. Despite some cultures and lifestyles sustainably managing natural resources and achieving good quality of life in all subregions, the aggregate ecological footprint is unsustainable and has increased two to three-fold since 1960. Patterns vary among countries and subregions {Table 2.24, Figure 2.36}; South America is the only subregion to retain a "reserve" of biocapacity; the others exceed nature's ability to renew its contributions to human quality of life (*well established*).

3. **The Americas' outstanding cultural diversity is highly threatened (*well established*).** While the region hosts 15 per cent of the world's languages, 61 per cent of this linguistic diversity (and associated cultures) is in trouble or dying {Table 2.1}. Major indigenous and local knowledge systems (e.g. in the central Andes and the Arctic) have shown their capacity to wisely manage territories based on particular values, technologies and practices, despite globalization processes {2.4} (*well established*). The Americas' diversity of cultures, including those arising from its immigrants, provide opportunities to develop sustainable practices and respect for nature.

4. **Food production is increasing in the Americas and is important for food security from local to global scales. Large-scale agriculture often replaces natural ecosystems with simpler ones, converting multiple nature's contributions to people and diverse livelihoods to one or many fewer nature's contributions to people or stakeholders.** Since 1960, crop production increased, except in the Caribbean. Natural habitat conversion and increased land productivity improved efforts to satisfy human demands for meat, crops and other commodities {2.2.1} at the expense of reduced biodiversity. This improved incomes for many rural people, while marginalizing others {e.g. Box 2.3}. Indigenous peoples and local communities have millennial polyculture and agroforestry systems that provide livelihoods, maintain biodiversity and shape landscapes {2.4} (*well established*).

5. **The region has largely overcome food insecurity, but disparities persist among countries and subregions. Hunger remains a problem and obesity is increasing.** Undernourishment affects more than 40 million people in Mesoamerica and South America, and 3.6 million face severe food insecurity in North America {2.3.1}. In Latin America and the Caribbean, undernourishment declined from 14.7 per cent of the population between 1990 and 1992 to 5.5 per cent between 2014 and 2016 {2.3.1}, while obesity greatly increased in all subregions (more than 30 per cent of adults in North America, and more than 20 per cent elsewhere) {2.3.1} (*well established*). Nutrition indicator improvements are associated with good economic performance, increased food/feed productivity {2.2.1}, but also social policies, and is not merely a result of per capita Gross Domestic Product (*established, but incomplete*).

6. **Regionally, freshwater is abundant, but areas affected by water scarcity are increasing. Water insecurity affects more than 50 per cent of the region's population (*well established*). Imports of "virtual water," in food and other commodities, from water-rich areas helps offset scarcity, but at the expense of environmental damage, like dead zones in the Gulf of Mexico from pollution and agrochemicals (*established but incomplete*).** Per capita freshwater availability decreased by around 50 per cent in 50 years due to population increases {2.2.10, Figure 2.19}. Management improvements provide more people access to clean water but may reduce water supply to ecosystems {Figure 2.29} (*well established*). Non-consumptive use by industry is the largest beneficiary in North America, while agriculture is first in other subregions. Mesoamerica and South America consume less than 10 per cent of the global water budget; North America uses around 15 to 20 per cent (more than three times regional per capita use), but its water withdrawals are declining {2.3.2} (*established but incomplete*).

7. **Energy produced from biodiversity (wood, biofuels) and hydropower increased regionally, contributing to energy security. Large-scale bio-energy production has trade-offs with food production and biodiversity, affecting local populations that depend on nature for livelihoods.** In about one third of countries, 100 per cent of people have access to electric power; in the rest at least 80 per cent have access (except one country). Only 11 countries depend on renewable energy (hydro, solar, wind and biomass-based energy) for more than 60 per cent of their electricity {Table 2.6}. Biofuels are increasingly important in South and North America's energy matrix with the United States of America and Brazil leading the world in ethanol production. Fuelwood is important for cooking, heating and lighting in localities with little or no access to electricity {2.3.3}. In North America, wood fuel is mostly for industrial use, whereas in South and Mesoamerica it is used in households {2.2.3} (*well established*).

8. **Human health depends directly and indirectly on nature. Biodiversity is a source of medicinal plants and animals, and chemodiversity with high potential economic value for pharmacological products.** Medicinal and aromatic plants in the Americas are valued at around $2 billion per year in 2016 dollars, and international trade is expanding {2.2.4} (*well established*). Experiencing nature and other non-material nature's contributions to people positively affect mental and physical health. Urban green spaces can decrease obesity in inner-city minority youth, and access to nature affects the recuperation of cancer survivors and the well-being of elderly disabled people {2.2.5, 2.3.5}. In addition, nature regulates pests and diseases and environmental quality. Ecosystem degradation, including biodiversity loss, can increase incidence of vector-borne maladies like Lyme disease, dengue fever and Zika virus {2.2.15}. Plus, over 8,000 children under the age of five years-old die annually in the Americas due to water-related diarrhea {2.2.11} (*well established*).

9. **Comprehensively evaluating how nature's contributions to people support good quality of life requires assessing the multiple values and value systems that underlie humans' relationships with nature (2.5.1, Table 2.15, Table 2.21)**. For example, food and feed can be evaluated relative to their biophysical metrics, like species richness or land cover occupied by biodiversity used as food {2.2.1}. However, edible biodiversity has health effects that can be positive (malnutrition has decreased in the last decades in the Americas) {2.3.1} or negative (agriculture-related pollution) {2.2.1}; and also relates to meaningful socio-cultural practices and nature-based livelihoods {2.2.1, 2.2.6, 2.3.5, 2.4} (*well established*).

10. **When economic values are assessed, the Americas' terrestrial nature contributions to people are equivalent to its Gross Domestic Product.** The regional monetary value of terrestrial ecosystem services is estimated at $24.3 trillion per year, similar to the region's $25.3 trillion Gross Domestic Product (2011 values). Brazil, the United States of America and Canada accounted for the largest monetary values ($6.8, $5.3 and $3.6 trillion per year, respectively). Antigua and Barbuda, The Bahamas, and Saint Vincent and the Grenadines had the highest values per area ($22, $21 and $18 thousand per hectares per year, respectively). Countries' size and the monetary value of specific ecosystem types cause these differences; biomes like coastal wetlands and rainforests having particularly high economic values {2.5.1, Table 2.22, 2.23} (*established, but incomplete*).

11. **Value plurality in the Americas shapes use, management and conservation of nature {2.1.2, 2.5}. Governance processes and tools, like prioritization and cost-benefit analysis, need to take into account multiple values {2.5.1, 2.7} (*established but incomplete*).** Doing so helps ensure nature´s contributions to people are prioritized in policy interventions to achieve specific sustainable development goals {Figure 2.37}. While it is clear that some material nature's contributions to people, like food and energy, are crucial to overcome Sustainable Development Goals 1 and 2 (no poverty and zero hunger), it is evident that the values plurality involved in quality of life from non-material nature's contributions to people (learning and inspiration), and transversal nature's contributions to people (maintenance of options) are equally important {2.7, Table 2.25}.

12. **Nature-based livelihoods, like fishers, farmers, loggers, ecotourism, depend on material nature's contributions to people (e.g. food) with high economic value that are quantified in national accounting, non-material nature's contributions to people that provide learning and experiences**

and support identities, as well as regulating nature's contributions to people that control disasters and disease. As the Americas' population becomes increasingly urban, trade-offs between city dwellers and rural residents mean that decision-making power rests with those who have a less direct relationship to nature for their livelihoods {2.1.1, 2.2.1.3, 2.3.1, 2.3.5, 2.5.1} (*well established*).

13. **While protected areas help ensure nature's contributions to people, nature's benefits also can be enhanced within human-dominated landscapes. Multifunctional landscapes contribute diverse nature's contributions to people and maintain long-term access**. Both preserving and restoring ecosystems maintain nature's contributions to people like pollination, pest control, water resources, erosion control and humans experience with nature. In North America, the fraction of protected land area (11.6 per cent) is less than the proportion of protected territorial marine waters (16.4 per cent). In Mesoamerica, the Carribbean and South America, the fraction of protected land area (23.5 per cent) exceeds the proportion of protected territorial marine waters (15.5 per cent) (*well established*). Indigenous land also can protect nature and constitutes 19.5 per cent, 11.1 per cent, 1.2 per cent of land in Mesoamerica, South and North America {2.2.8} (*established but incomplete*).

14. **While poverty rates have decreased since the 1990s, large populations, particularly in Mesoamerica, South America and the Caribbean are still vulnerable. Social inequality is high; 10 of the world's 15 countries with the most unequally income distribution are in the Americas {2.3.5}.** Data indicate that South America has the most socioenvironmental conflicts (*inconclusive*). Even when nations enshrine citizens' rights to nature and nature's contributions to people, like clean water, little information exists regarding trends and status of actual access and benefits sharing for different social actors {2.5} (*established but incomplete*).

15. **Loss and degradation of wetlands and forests have reduced nature's contributions to people for climate regulation and adaptation to hazardous and extreme events (*established but incomplete*).** Carbon stored in wetland soils and forests is critical for climate regulation {2.2.9} (*well established*). Wetlands reduce disaster risk and cleanup costs (e.g. the United States of America coastal wetlands reduced storm damage by around $625 million during hurricane Sandy) (*established but incomplete*). Peak flood flows are moderated by the presence of riparian wetlands, floodplains, lakes and ponds. Natural vegetation also moderates the chances of avalanches and landslides {2.2.12} (*established but incomplete*).

16. **Information gaps detected during this assessment include: i) social data are generally collected at the political scale, while ecological information is taken at the ecosystem or biome levels, impeding integration and comparison, ii) some political entities are under-represented or absent from global country-level databases (e.g. Greenland), iii) relative absence of long-term data, particularly for some regulating and non-material nature's contributions to people, and iv) relative absence of multiple valuations and trade-off analysis of human-nature relationships {2.8}.**

2.1 Introduction

Humans and nature are inextricably and intricately linked (see Chapter 1**,** Figure 1.4). Human well-being depends upon nature in ways that are direct or indirect, simple or complex, and reciprocal or uni-directional (Pascual et al., 2017). In the Americas region, the strength and intensity of these human-nature relationships vary over time (e.g. within and between generations), between subregions (e.g. North America, Mesoamerica, Caribbean, South America) and among different social groups (e.g. primary and secondary users of nature) (MEA, 2005). In addition, the ways we conceive, study, value, and manage these links are variable, depending on one's worldviews and value systems; therefore, appreciating the different ways that nature is valued broadens our understanding of the benefits it provides. While it is increasingly understood that human-nature connections are ubiquitous and important, however, their breadth and nuance make them difficult to incorporate into political and technical decision-making processes (see Chapter 6), and more fully describing and quantifying nature's contributions to people (NCP) become crucial for motivating, orienting and justifying policy development and management actions (Díaz et al., 2015).

The utilitarian assumptions that underlie the ecosystem services evaluations of human-nature relations are not sufficiently broad to ensure a full understanding of how peoples around the world interact with and benefit from nature. It provides a useful framework, however, for assessing the importance of ecosystems and has become a core concept in wide use by many countries and organizations worldwide, providing valuable language and tools for common discussion and understanding (Laterra et al., 2011; Seppelt et al., 2011; Balvanera et al., 2012; Pascual et al., 2017). The Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) has introduced complementary concepts like NCP and quality of life, which help interpret the significance of the globe's biodiversity to diverse people and their understandings of well-being. Therefore, the IPBES conceptual framework (Díaz et al., 2015; Pascual et al., 2017) employs two strategies: 1) it builds upon the ecosystem services paradigm to assess how NCP affect human well-being (section 2.1.1), and 2) it recognizes and seeks to incorporate multiple social actors, who hold diverse values and knowledge systems in the appraisal of both NCP and quality of life (section 2.1.2).

In this context, Chapter 2 of the IPBES Americas Regional and Subregional Assessment reflects a shift in emphasis from ecosystem services to NCP as a way to more explicitly highlight the role of nature in supporting quality of life and broaden our appreciation to be more inclusive of worldview and value plurality (Pascual et al., 2017). Current information on the values of nature is largely a result of the academic history of the ecosystem services concept. As this approach has increased exponentially throughout the world (Seppelt et al., 2011) and the Americas (Laterra et al., 2011; Balvanera et al., 2012), most studies have concentrated on two aspects of human-nature interactions: (a) addressing how ecosystem properties (e.g. biotic assemblages) and functions (e.g. biogeochemical cycles) are used by humans and human institutions (i.e. managed) to produce "final services" (*sensu* Fisher et al., 2009), and (b) the economic valuation of these benefits to human society. This chapter reflects these approaches to valuation and also seeks to enhance them with a values plurality strategy that recognizes other valuation methodologies (section 2.5.1).

2.1.1 The diversity of nature's contributions to people and links to quality of life

Nature's contributions to people encompass a broad array of material, non-material and regulating biophysical benefits to humans (geophysical benefits are not addressed here) (see Chapter 1, Table 1.1) and underlie key components of human well-being that define a good quality of life (Daily, 1997; MA, 2005). Specifically, the Americas' biological and ecosystem diversity make material contributions, in the form of food, fiber, energy, water, materials and assistance (fiber, dyes, cloth, decorations, labor, transportation, pets), medicine, and biochemical and genetic resources, to the security of livelihoods and energy. Regulating contributions of nature, including habitat and soil creation and maintenance, pollination and seed dispersal, and the control of diseases, pests, natural disasters, climate, air and water quality, and ocean acidification, strongly affect human health and the securities of food and water. Non-material contributions, such as learning and inspiration, psychological and physical experiences, support for identities, and the maintenance of options, are key to sustaining place-based livelihoods (or ways of living) and cultural continuity. In turn, these NCP are constituted from the region's high biological and ecosystem diversity, which provide such attributes and functions as habitat for species, biomass production, carbon storage, or nutrient uptake (see Chapter 3), and as such, biodiversity and ecosystems are embedded in the ecosystem services that produce NCP (Worm et al., 2006).

While it is critical to identify and account for the specificity of place, culture and community in any assessment, studies have shown that biological and cultural diversity and extinction risk follow similar geographic patterns at a global scale (Collard & Foley, 2002; Sutherland, 2003). The Americas present a unique scenario for studying these patterns, though, and their implication for human well-being. First, the region displays a greater latitudinal range than any other (~80°N-56°S). Furthermore, it hosts not only a great diversity of species and biomes (see Chapter 3), but also numerous cultures. Indeed, the Americas have the highest cultural diversity of any region (Collard & Foley, 2002), but many of these human groups are small. Only about 6.5% of region's total population of approximately 1 billion is categorized as indigenous, but in the Mesoamerica subregion the percentage increases to 16.9% of the population (see Chapter 1, Table 1.3). At the same time, the Americas host ~15% of the world's living languages, but this linguistic diversity is highly threatened. Globally, the Americas is the region with the greatest number of dying languages (n=341, Table 2.1), and overall, ~61% of the Americas' languages are considered "in trouble" or "dying" (Simons & Fennig, 2017), which is greater than the percentage of biological species in the equivalent threatened status (see Chapter 3).

Table 2.1. Languages from the Americas per subregion, indicating conservation status.

	Total Languages	Languages in Trouble	Dying Languages	Total %Threatened
North America	256	80	157	92.6
Mesoamerica	326	91	42	40.8
Caribbean	23	3	3	26.1
South America	456	133	139	59.6
Americas Region	**1,061**	**307**	**341**	**61.1**
Global	7,099	1,547	920	34.8
Americas as % of Global	14.9	19.8	37.1	26.3
Source: Simons & Fennig (2017).				

The interaction of the Americas' social and ecological diversity provides multiple, often unapparent, ways for humans to relate to nature. For example, the domestication of plants in the Amazon in the pre-Colombian era continues to structure the vegetation composition of the modern forest (Levis et al., 2017). However, the scholarship on the NCP-quality of life relationship does not fully address this complexity. The number of studies on the benefits people receive from nature has a bias towards Western developed nations; one major review found that 79% of such publications were from North America and Europe with none from South America and Africa (Keniger et al., 2013). There are also gaps in the information available on different biomes or valuation methodologies. For example, a review of the effects of conservation interventions on human well-being in countries that were not members of the Organization for Economic Cooperation and Development found that among 1,043 studies evaluated, there was a clear emphasis on research in forested ecosystems and the material and economic benefits of conservation and the effects on governance (McKinnon et al., 2016). Other aspects of well-being, such as health and livelihoods have been less studied, and overall, only 9% of publications used quantitative methods. Therefore, although there is clear consensus in the literature that NCP are important for human well-being, it is often difficult to discern the status and trends in the ways the constituent parts interact.

At the same time, while long-term quantitative information is sometimes lacking, insights can be gained by examining the qualitative ways that NCP and human well-being are related, including a mechanistic understanding of how knowing, perceiving, interacting with and living in nature affects well-being (Russell et al., 2013). This chapter seeks to highlight these relationships and the particular values at stake (see Table 2.1 in the document IPBES/3/INF/7 "Preliminary guide regarding diverse conceptualization of multiple values of nature and its benefits, including biodiversity and ecosystem functions and services", and also

Table 2.21 in section 2.5.1 of this chapter). For example, there is a well-established and plural human-nature relationship between salmon and various indigenous peoples and local communities in northwest North America. On the one hand, salmon are used to help satisfy material needs of direct beneficiaries (e.g. meat) and also represent an economic resource for indirect users. At the same time, some groups value salmon for their aesthetic/spiritual properties that contribute to non-material NCP (NRC, 1996). Furthermore, the value of an NCP to quality of life can vary over timing of its delivery. For example, in the case of habitat conversion and pesticide application to increase crop yields, it is necessary to also account for the concomitant reduction in pollinators that ultimately can jeopardize food security in the medium- to long-term (IPBES, 2016).

Consequently, IPBES' current assessments of the NCP-quality of life relationship provide a way to systematize and monitor how human development and environmental conservation relate to one another and how different values and timescales are linked. This effort also advances and complements other international programs like the United Nations (UN) Sustainable Development Goals (SDG) and the Convention on Biological Diversity's (CBD) Aichi targets, which share an expansive understanding of the human-nature relationship and an emphasis on developing quantitative measures that allow implementation in policy- and decision-making.

2.1.2 Understanding stakeholder, value and knowledge system diversity in the human-nature relationship and its effect on quality of life

Evaluating the relationships that humans develop with nature requires taking into account the diversity of stakeholders, their values and knowledge systems. For instance, the distribution of benefits and disservices varies within and between social groups, whereby asymmetries in access to nature that are based on gender, age, social role or status, and other characteristics affect outcomes regarding human well-being. Stakeholders, in turn, may define their well-being based on group values, determined as a function of their social role or way of life (e.g. farmers, decision-makers or local residents), or based on their personal interests (e.g. users, providers or intermediaries of nature's contributions to people). Consequently, social valuation of nature and its ecosystem benefits and services varies, depending on individual stakeholder traits, such as socio-economic status and literacy levels (e.g. factors that affect willingness to pay for ecosystem services, Silva et al., 2016) and also on broader worldviews that are shared by specific cultures or social groups (e.g. Andean cosmology, which considers the human-nature relationship reciprocal, Zenteno-Brun, 2009) .

Furthermore, the loss of ecosystem services does not impact communities equally; often losses are felt disproportionately by marginalized peoples (e.g. developing nations, lower income communities, and ethnic groups with more direct traditional ties to nature) (MEA, 2005). Moreover, powerful stakeholders, such as large industry and government agencies, have greater capacity to impose their worldview and values upon others by more heavily influencing management decisions, compared to less powerful social actors, such as small-scale farmers or indigenous hunters, whose quality of life depends more directly upon local ecosystems (e.g. Darvill & Lindo, 2016). Indeed, different groups may not only have divergent power, uses and interests, but they also may define the very concepts of nature and quality of life based on different knowledge systems (IPBES/4/INF/13).

Balancing the contested needs, demands and conceptualizations of nature proves increasingly difficult, as species and ecosystems are shared across a greater number of stakeholders and jurisdictions (i.e. telecoupling). When conflicts between social groups arise, it is important that these also be understood from the standpoint of stakeholder value and knowledge diversity, which must be incorporated for successful management (Mouchet et al., 2014). For example, when confronted with the possibility of building a dam on the Upper Peace River in British Columbia, Canada, environmentalists, government officials and recreationists placed lower value on provisioning ecosystem services than First Nations peoples, hunter/anglers and agriculturalists. In contrast, cultural ecosystem services, such as the aesthetics and beauty of landscapes, landscapes for sense-of-place, and recreation were consistently ranked highly across all groups (Darvill & Lindo, 2016). It is important, therefore, to recognize the multiple ways of understanding nature for decision-makers to incorporate the breadth of values at stake before conflicts occur (Jones-Walters & Cil, 2011; Klain & Chan, 2012). By elucidating the stakeholders, values, and knowledge systems at play, programs can determine the underlying preferences and motivations that characterize social-ecological interactions and the subsequent valuation of ecosystem services (e.g. Silva et al., 2016), and better management plans can avoid conflicts by not taking decisions that create asymmetries in the availability of ecosystem services or that unwittingly prioritize one stakeholder or value over others (Howe et al., 2014).

In the following sections, we assess the status and trends of NCP and how these ecosystem services impact human well-being in the Americas. This assessment uses and expands upon the ecosystem services paradigm (Ehrlich & Mooney, 1983), which rose to prominence in the ecological sciences as part of a broader academic and intergovernmental understanding that human societies (including economies) are bound by ecological constraints (Meadows et al., 1972; Brundtland et al., 1987). Subsequently, the concept was developed as a central element in the fields of economics and natural resource management (Gómez-Baggethun et al., 2010), and today it is found expressed in policy instruments across the Americas (e.g. native forestry laws in Argentina #26,331, Bolivia #1,700, Brazil #11,284 and Chile #20,283) and international initiatives (e.g. World Bank's Wealth Accounting and the Valuation of Ecosystem Services and The Economics of Ecosystems and Biodiversity).

The IPBES approach aims to broaden the valuation of nature by explicitly incorporating stakeholders, values and knowledge systems (Pascual et al., 2017). This socio-cultural valuation approach (see Scholte et al., 2015) facilitates a broader understanding that integrates insights from environmental philosophy and ethics (intrinsic, instrumental and relational values, Rolston, 1986; Callicott, 1989; Chan et al., 2016) and environmental social science disciplines, such as sociology, social psychology and anthropology (Keen et al., 2005; Clayton & Myers, 2009; Steg et al., 2013; Díaz et al., 2015), with an explicit recognition and validation of the values and knowledge held by indigenous peoples and local communities. In keeping with this approach to values, it is equally important to recognize from where one speaks, and this chapter (and the entire *Americas Assessment*) was developed primarily by academic scientists with a natural science education and background. However, by making this fact explicit and applying the integrated assessment methodologies developed by IPBES, such limitations can be addressed, but should never be overlooked in the interpretation and analysis of findings.

2.2 Status and trends of nature's contribution to people in the Americas

In the following sub-sections we present: (i) data showing the status and trends of NCP in the Americas and its subregions (North America, Mesoamerica, the Caribbean and South America); (ii) the contributions of each category of NCP to quality of life (i.e. human well-being); (iii) select case studies to demonstrate relevance, observed differences between subregions, and differences in cultural values or trends in a particular variable; and (iv) where appropriate, a brief description of drivers affecting the NCP and it links to well-being (see Chapter 4 for a quantitative discussion on drivers and impacts). The section is organized by material, non-material and regulating NCP (see Table 1.4 in Chapter 1).

2.2.1 Food and feed

Agriculture is a dominant form of land management globally, and agricultural ecosystems cover nearly 40% of the Earth's terrestrial surface area. According to the Food and Agriculture Organization of the United Nations (2014a) most farms are owned by the families that work them; they tend to be small and found in rural areas of developing countries. Many small family producers are poor, food insecure and have limited access to markets and services. Despite this, they cultivate their own land and produce food for a substantial proportion of the world's population. In addition to agriculture, they engage in many other (often informal) economic activities to supplement their reduced incomes. Agricultural ecosystems are managed by people mainly to meet food, fiber (section 2.2.2) and fuel needs (section 2.2.3) (FAO, 2014a). An extensive body of evidence shows that agricultural investment is one of the most important and effective strategies for economic growth and poverty reduction in rural areas (FAO, 2015). Continuing growth of populations and increasing consumption per capita means that the global demand for food will increase for at least another half-century. The competition for land, water, and energy, in addition to the overexploitation of fisheries, will affect humans' ability to produce food and contribute to the urgent requirement to reduce the impact of the food system on the environment and other NCP. Plus, the effects of climate change are a further threat (Alston et al., 2000, see Chapter 4 for more details).

2.2.1.1 Crops

The Americas play a key role in the sources and production of crops in the world's economy, showing an increase in production rates for some commodities that is higher than the global trend. The Americas provided about 17% of global production in oil crops and 10% of coarse grain (primarily corn). The region is also a net exporter of sugar and honey, with exports more than doubling between 2000 and 2011 (FAO, 2014a). This growth is due to the tremendous increase in exports from Brazil (from 6.5 million tons of sugar in 2000 to 25.5 million tons in 2011), making it the world's largest sugar exporter.

The average agricultural productivity in the Americas, measured as the real agricultural aggregate value per farm worker was $3,070 from 2000 to 2009. This regional average is much lower than specific subregions or countries. For example, in Canada it is $42,965 per farm worker (The World Bank, 2012). The increase of the real agricultural aggregate value of the South American subregion was an extraordinary 10.8% in 2009, almost three percentage points above the subregional Gross Domestic Product (GDP) increase (see Chapter 1), primarily due to record wheat yields in Brazil and Argentina and corn in Argentina (CEPAL/FAO/IICA, 2012).

Crop production increased overall between 1961-2013 (Figure 2.1). In the Caribbean, however, where sugar had been the most important agricultural commodity, production significantly decreased (FAO, 2014a). This was in part a result of the USA economic blockade of Cuba, and of more competitive sugar production in other regions (FAO, 1997). In Mesoamerica, some crop decreases were mainly due to changes in trade policies with a tendency to deregulate domestic markets and reduce trade barriers. On the other hand, North America registers a constant growth for soybean and wheat crop production, while other crops remained stable. In South America, soybean and corn production increased substantially in recent years. By 2014, the area allocated per subregion for cereal cultivation stands at >127 million hectares, where North America accounts for >50% of the region's total and annual growth was only observed for cereal production in Mesoamerica and the Caribbean (Figure 2.1).

Figure 2 1 A Production for 1960–2016 of the 10 crops most produced in each subregion. Source: FAO (2017). FAOSTAT Statistics Database. http://www.fao.org/faostat/en/#data/QC. Date accessed: August 27, 2017.

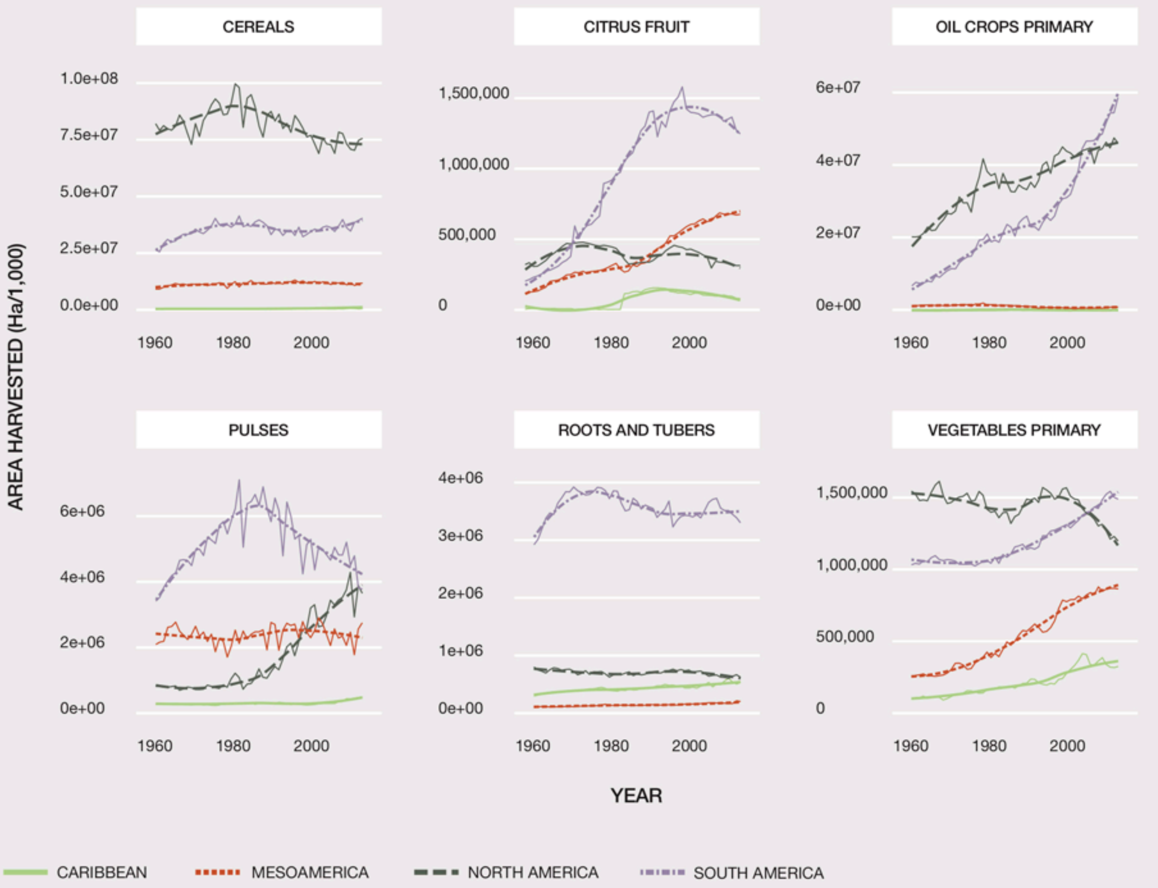
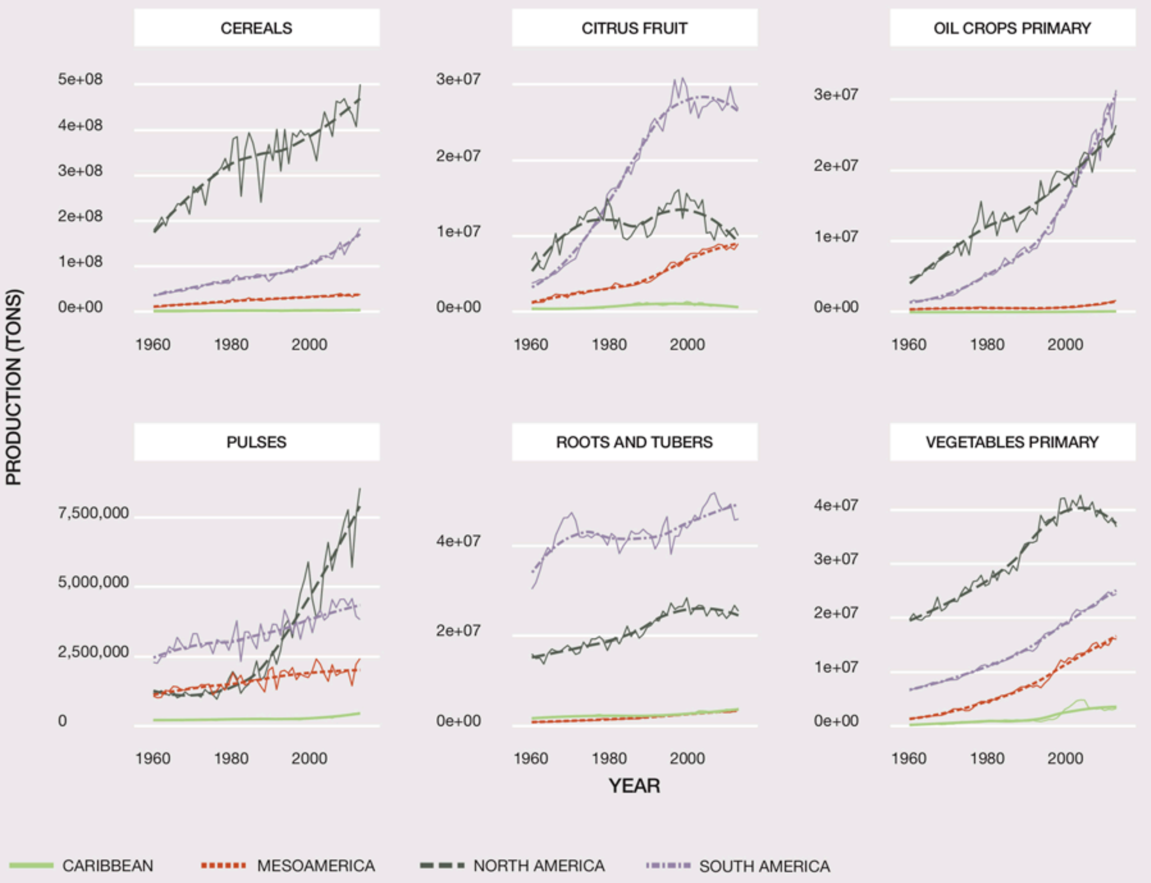


Figure 2 1 B Production for 1960–2016 of the 10 crops most produced in each subregion. Source: FAO (2017). FAOSTAT Statistics Database. http://www.fao.org/faostat/en/#data/QC. Date accessed: August 27, 2017.



Overall, the Americas has positioned itself well in the international market of agricultural goods, and the export of agricultural products has increased dramatically for all subregions in the Americas, except the Caribbean. (CEPAL/FAO/IICA 2012). Without considering the type of crops, a comparative view shows patterns with export and import values for the Americas (Figure 2.2). The Caribbean's decrease in exports is the result of reductions in sugar cane export since 1989, while the export of soybeans was the most important crop commodity from South America (Figure 2.3). Throughout the past 50 years, corn, soybean and wheat showed the highest export values for North America, and bananas, vegetables and sugar for Mesoamerica.

Figure 2 2 Exports and import trends of agricultural products 1960–2013 of the Americas. Source: FAO (2017). FAOSTAT Statistics Database. http://www.fao.org/faostat/en/#data/TP. Date accessed: August 27, 2017.

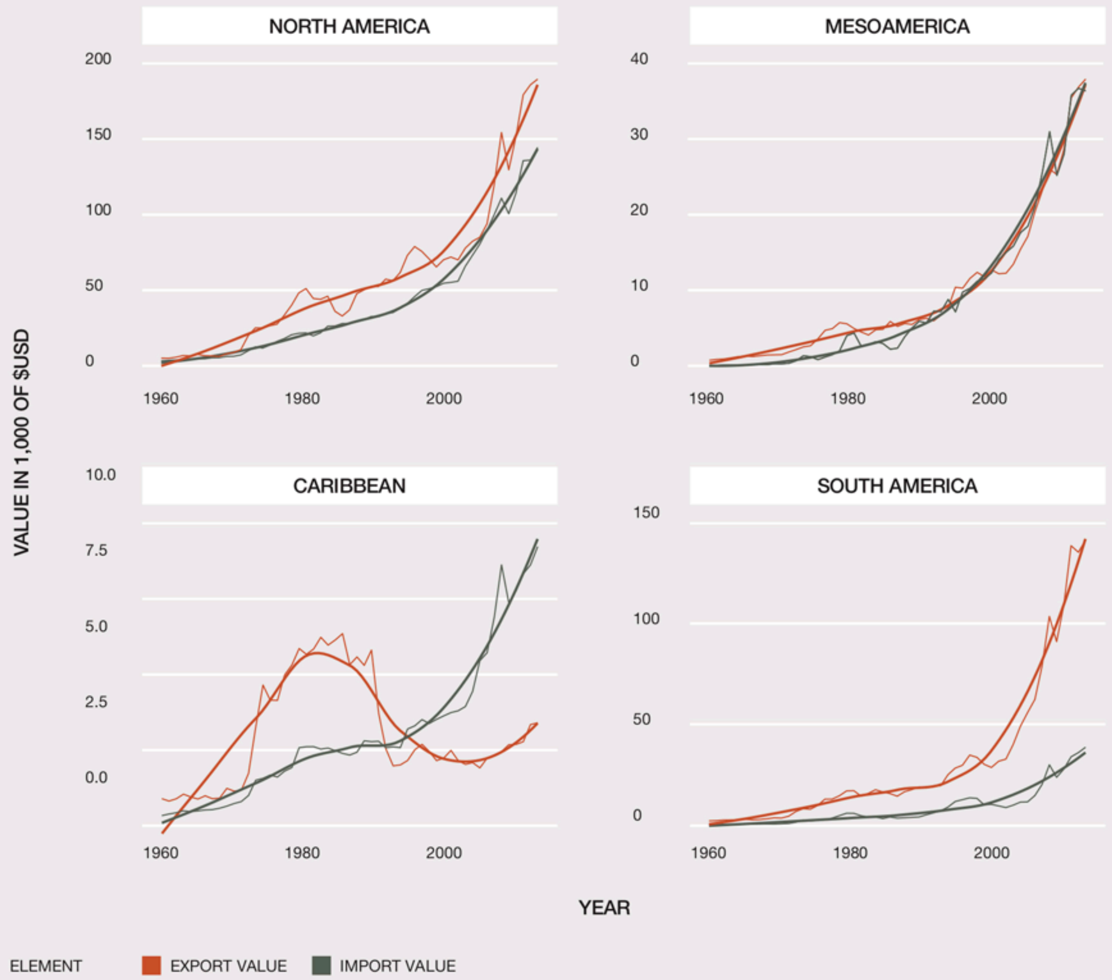
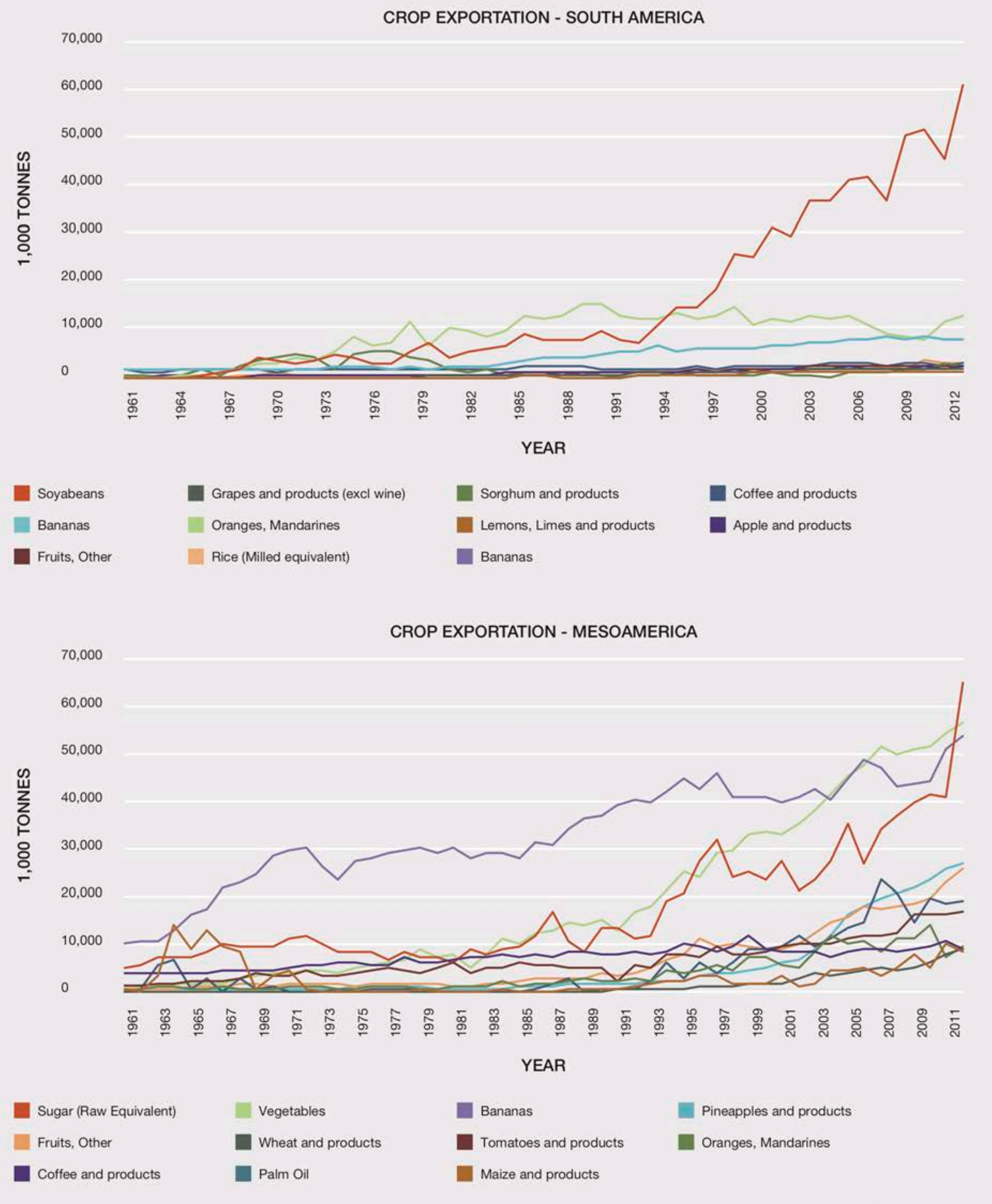
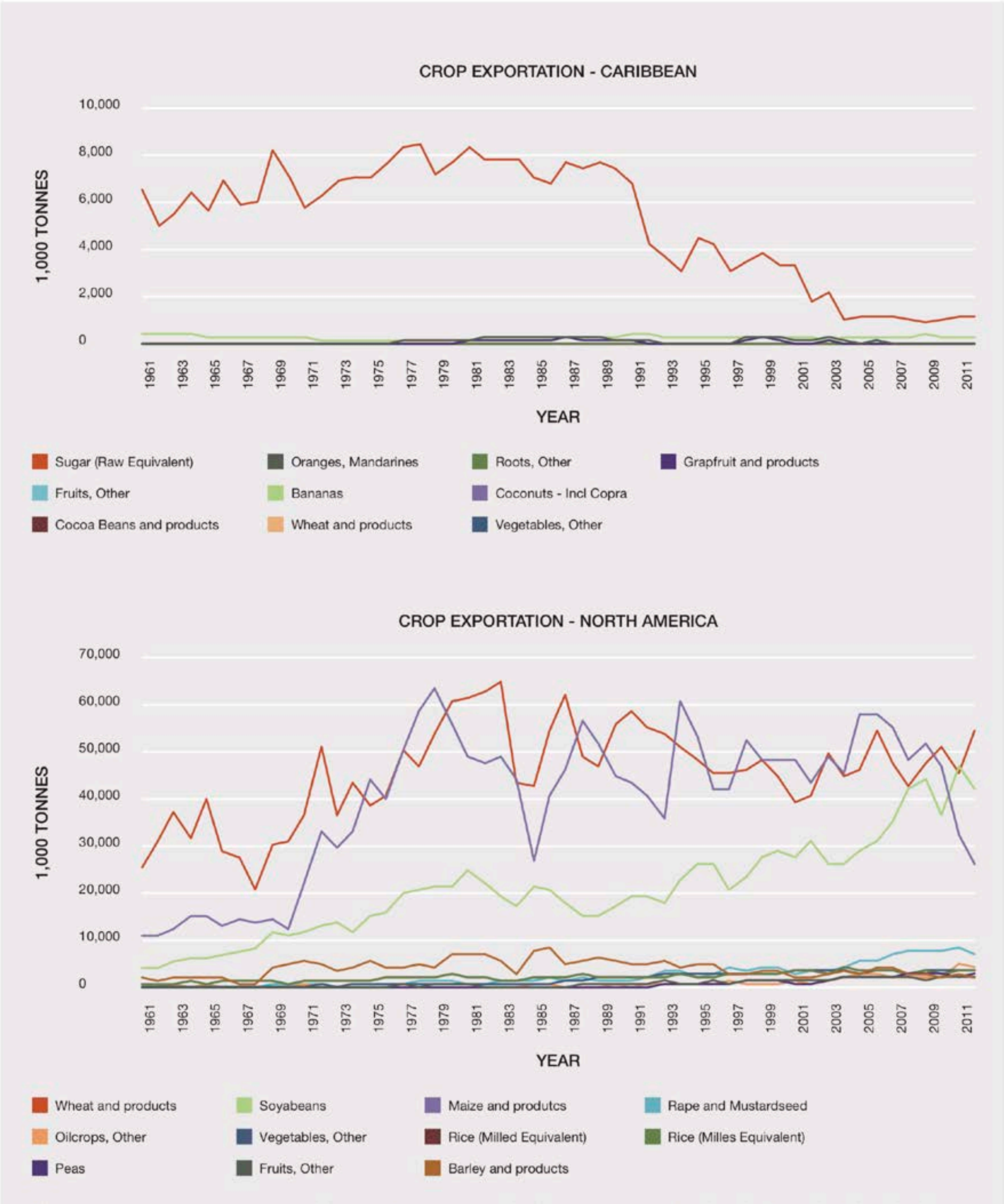


Figure 2 3 **Export trends of crops from the Americas, 1961–2013. Source: FAO (2017). FAOSTAT Statistics Database. http://www.fao.org/faostat/en/#data/QC. Data accessed: February, 14, 2017.**





With agricultural industrialization and the increasing use of commercially-distributed seeds, native cultivars or breeds that are important for the long-term food security of American people are increasingly at risk of extinction. For example, the availability of lands under adequate climatic and soil conditions restricts crop production, and irrigation will become increasingly important in many subregions as agricultural land use has expanded (Fischer et al., 2002). Based on currently available soil, terrain, and climatic data, the Global Agro-ecologial Zones assessment estimates there are 10.5 billion hectaresof agricultural land globally and 4.2 billion hectares for the Americas (CEPAL/FAO/IICA, 2012). The increasing demand for crops, however is evidenced by crop importation in all subregions (Figure 2.2). For example, corn imports increased in all subregions, while wheat increased in all but the North American subregion.

The Americas region has a high diversity of useful plants that historically have been naturalized and diversified creating crops, cultivars, and varieties based on properties such as weight and nutrition, and value for local communities (section 2.4). Staple foods like potatoes, corn, pepper, many varieties of beans, and tomatoes, were developed as food products by people long before European settlement; these traditions remain alive, especially in the farmers of indigenous descent from Mexico to Argentina (FAO, 2014a). The risk of extinction of native cultivars or breeds is a concern for the long-term food security of people in the Americas; the largest contributor to the loss of wild relatives of today's crop species is the destruction of natural landscapes. The loss of genetic diversity is also a major concern; according to FAO (1999), since the 1900s some 75% of plant genetic diversity has been lost as farmers worldwide have abandoned their multiple local varieties and landraces for genetically-uniform, high-yielding varieties. In addition, wild populations that can be genetically stronger and with better resistance to pests have also disappeared or are no longer used to improve cultivated plants.

An example is the cultivation of corn (*Zea mays*). The primary gene pool includes maize and teosinte (*Zea mays* subsp. *parviglumis*), with which maize hybridizes rapidly and produces fertile progeny. The secondary gene pool includes *Tripsacum* species (approximately 16), some of which are at risk of extinction, and the variability among native maize breeds (about 300 have been identified) exceeds that of any other crop (GCDT, 2007). A second case is the cassava (*Manihot esculenta*), which is important not only for the Americas region, but it is essential for food security in most parts of Africa. The gene pool is composed of this species and between 70 and 100 wild *Manihot* species, such as *M. flabellifolia* and *M. peruviana;* the wild primary sources of genes and genetic combinations of the new varieties are difficult to use and preserve (Allem et al., 2001). A third example is an Andean tuber, the potato (*Solanum tuberosum*). A recent study on the effect of climate change predicts that between 7 and 13 out of a total of 108 wild potato species may be driven to extinction (Jarvis et al. 2008) and there are reports on the vulnerability of *Solanum phureja*, a diploid species grown in the Andean zone (Terrazas et al., 2008).

2.2.1.2 Livestock

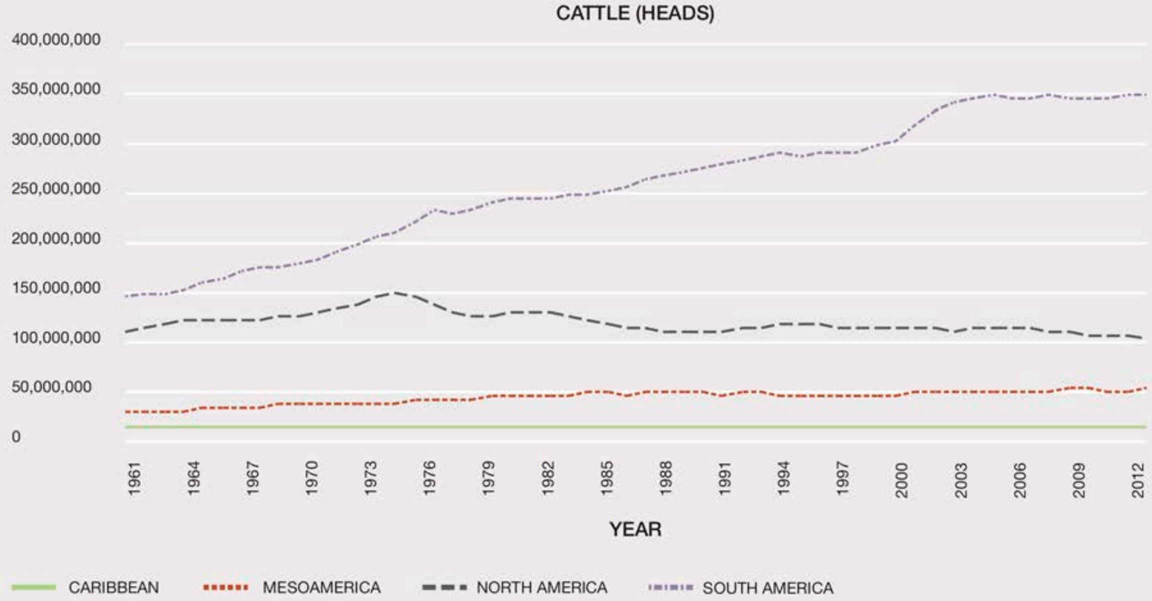
Livestock production is one of the fastest-growing agricultural sectors, especially in developing countries (The World Bank, 2009), with large contributions to local, regional and global economies. Livestock production systems provide several benefits including food for direct consumption or for commercial use on local, regional, national or global markets. The most important marketable products are meat, milk and eggs, with North and South America clearly leading in terms of export values (Table 2.2). Importantly, crop production in some regions serves as feed and fodder for livestock in other parts of the world: Argentina (31,200,000 Metric tons-MT), Brazil (15,500,000 MT) and the USA (11,068,000 MT) are the leading countries in exports of soybean meal, with most of the production going to the European Union (19,500,000 MT), followed by Asian countries (estimates for 2017; source: www.indexmundi.com, based on United States Department of Agriculture data).

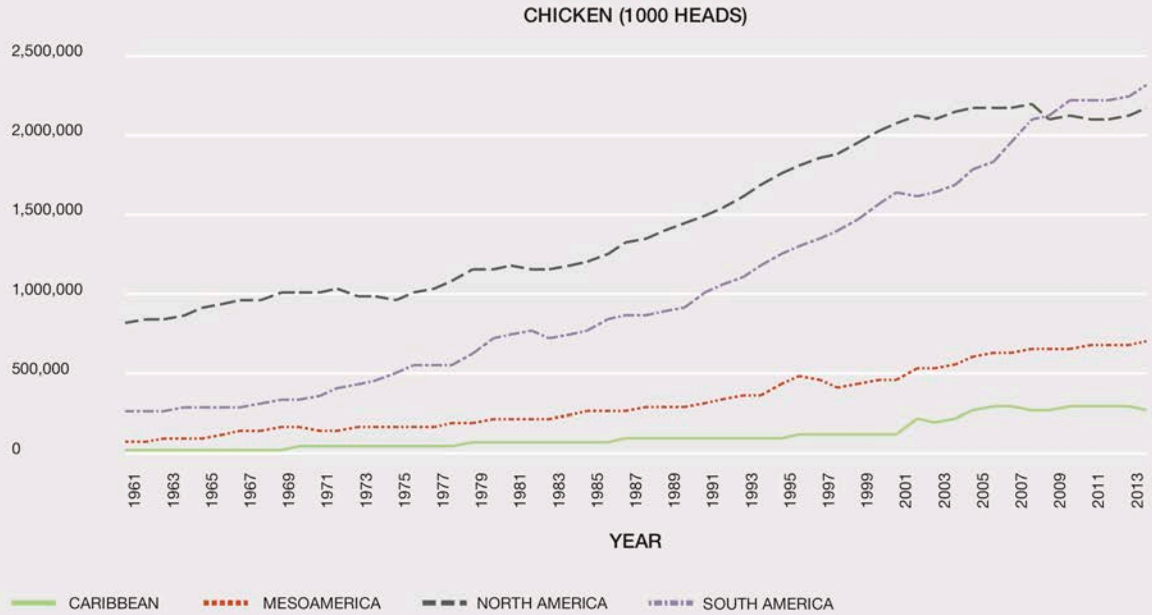
Table 2.2. Livestock trade monetary values in the Americas.

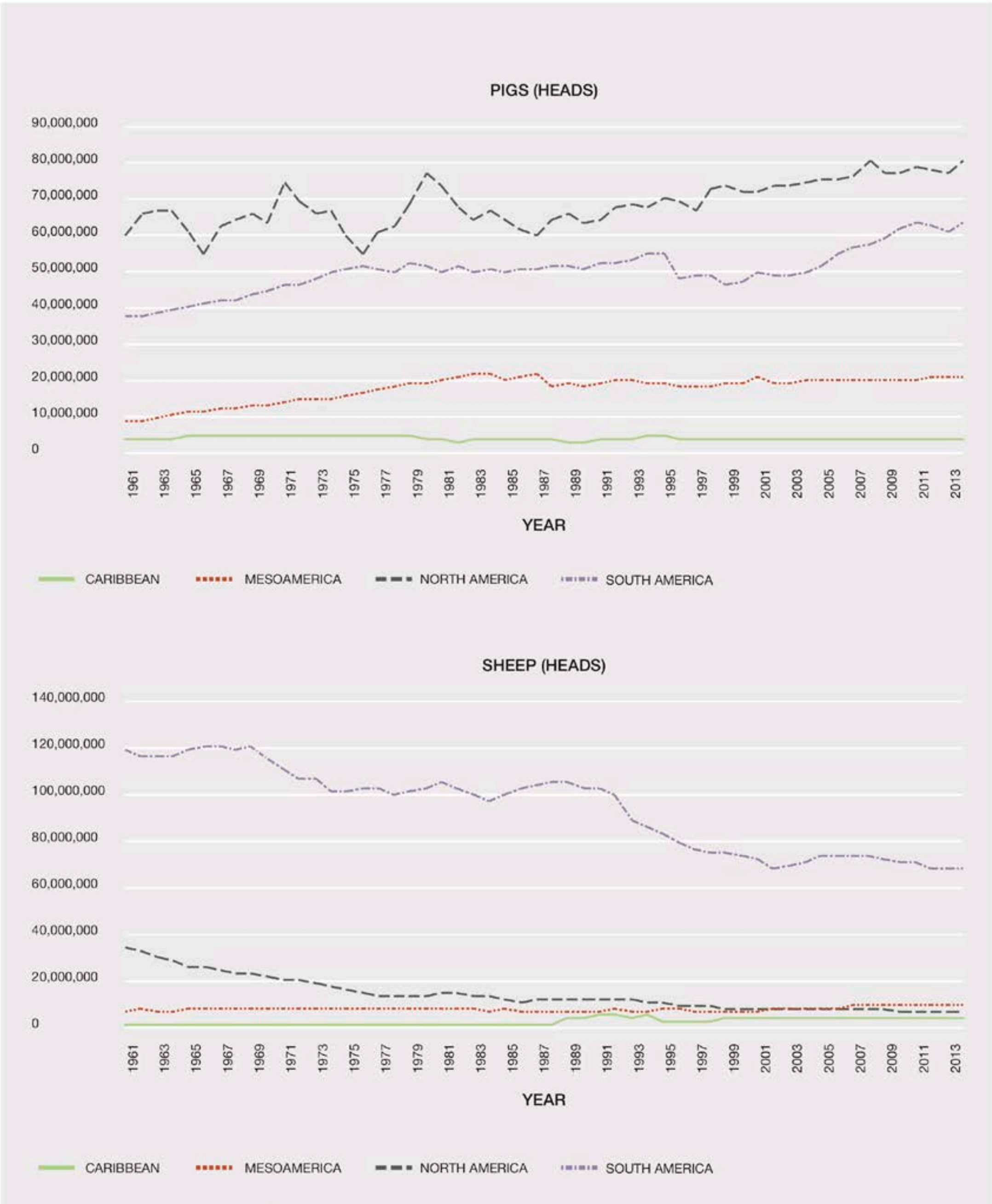
Item	Export Value (US $ in millions)				Import Value (US $ in millions)			
	CA	MA	NA	SA	CA	MA	NA	SA
Eggs	21.1	283.5	5815.8	1149.4	903.2	1206.3	2015.8	1155.4
Meat Bovine	375.4	11690.2	91526.1	96448.0	3042.1	17610.0	99398.5	20061.2
Meat Swine	7.5	4015.3	83110.3	18337.4	1556.7	10308.7	26719.0	3274.4
Meat Poultry	59.2	322.3	65792.7	72244.6	6574.8	11883.9	7066.9	6208.9
Meat Sheep	5.8	10.7	431.7	2531.2	854.0	1078.7	11111.9	558.9
Milk	206.0	2611.1	31350.9	17362.7	9886.7	21703.2	7992.1	23747.1
Total	**674.9**	**18.933.0**	**278.027.3**	**208.073.3**	**22.817.6**	**63.790.7**	**154.304.2**	**55.005.9**
Caribbean (CA), Mesoamerica (MA), North America (NA), South America (SA). Source: FAOStat (2015)								

Depending on the methods used, livestock production can have positive or negative effects on natural-resources, public health (e.g. through contaminated water supplies), and social equity (The World Bank, 2009). Drivers for production and choice of production systems are population growth (higher food demand globally), urbanization (with infrastructure improvements, e.g. cold chains) and income growth (Thornton, 2010). Rising demands lead to the transformation of natural ecosystems into lands for food (Alkemade et al., 2013). Global demand together with rising productivity has led to an increase in livestock, particularly in South America (Figure 2.4), and the Americas are predicted to have the largest expansion of rangeland area between 2000 - 2030 (Alkemade et al., 2013). Importantly, unlike crop species in the Americas, livestock production depends almost exclusively on domesticated animals originally exotic to the Americas that were introduced during European colonization of the region more than 400 years ago. Exceptions are camelids (llama and alpaca, domesticated from guanaco {*Lama guanicoe*} and vicuña {*Vicugna vicugna*}, respectively) and some small rodents (e.g. guinea pig {*Cavia porcellus*}) in South America. However, the total number of camelids in South America in 2014 represent only 0.25% of total number of cattle.

Figure 2 4 Production of the most important livestock in the Americas, 1961–2012. Source: Data from FAOSTAT (Production – Live Animals – Stocks) http://www.fao.org/faostat/en/#data/QA. Last accessed on Feruary 11, 2018.







Natural grasslands comprise almost 30% of the Americas (White et al., 2000), dominating the landscape in a diversity of regions including the Patagonia steppe (Argentina), the Pampas grasslands (northern Argentina, Uruguay, southern Brazil) and the North American prairie (USA, Canada). Here, sustainable grazing by livestock can be an economic activity that does not deplete the resource, in contrast to row crop agricultural land use (Herendeen & Wildermuth, 2002). This is because these natural grasslands evolved under the presence of large herbivores, whose role is now at least partly taken by domesticated animals. Natural rangelands provide many other benefits than those related directly to livestock production. For example, natural grassland conservation contributes to carbon storage in soil, prevents soil erosion, preserves groundwater quality and quantity, conserves native biodiversity and sustains local landscapes (Tanaka et al., 2011).

In other biomes – the most prominent example being Brazil’s Amazon forest– livestock grazing occurs after complete destruction of the natural ecosystems, or livestock may be raised in confined systems, based on feed produced in the place of natural systems, such as soybeans. Rarely does economic data on livestock distinguish the different production systems, which is a problem for measuring the relative degree of benefits and impacts regarding nature. Indeed, detailed sub-national characterization of livestock production, trends, and changes in relation with the ecological features of the area in question is necessary for an evaluation of the impacts of livestock production.

From a subregional perspective, in 2015, the livestock industry in the USA contributed over $60 billion to the national economy (USDA, 2017), clearly showing the importance of livestock production across biomes and production systems, including in small-scale systems, such has in the Great Plains where more than 85% of farms and ranches had less than 100 head of cattle (Mitchell, 2000). Trends of livestock numbers over the past decades in North America vary; the numbers of cattle and sheep are decreasing, and pigs and especially chicken – i.e., livestock raised mostly in confined systems – are increasing. For the near future, meat production is expected to increase for pork and chicken, meaning an intensification of production.

In South America, the products derived from natural grasslands are an important basis for regional or national economies. In 2013, the beef cattle population in southern Brazilian grasslands amounted to 13,592,000 heads (Souza et al., 2014) and just to the south Uruguay held 11,800,000 heads of cattle in 2014 (USDA, 2014). Together, Brazil and Argentina produced 19.6% of global beef production in 2015 (FAS/USDA). In Uruguay, where cattle are produced predominantly on natural rangelands, 240,150 tons of beef were exported with a total value of $1.3 billion in 2013 (USDA, 2014).

Even though productivity and thus economic returns may be lower, grazing is also important in some tropical savannas, such as in central Brazil (Carvalho, 2014) or the Llanos in northern South America (White & Thompson, 1955), where it presents a type of land use compatible with conservation of natural ecosystems and also of cultural significance. On the other hand, deforestation to gain land for other land uses, including livestock production, remains a major issue in tropical forest regions (see Chapter 4). After a clear reduction beginning in 2004, deforestation rates are on the rise again in Brazil, to cite just one example.

Livestock production in Mesoamerica is characterized by extensive grazing systems and mixed crop-livestock farming systems (Hellin et al., 2013), which are a key contributor to national food production and rural livelihoods, and play a central role in food security and economic stability (sections 2.3.1 and 2.3.5). In northern Mexico, where livestock grazing occurs in arid ecosystems and intensive feedlots, a variety of supplementary feeds are used. Plus, livestock expansion by converting forests to pasture is projected to be nearly insignificant in Costa Rica while impacting a considerable portion of Nicaragua's and Panama's forest cover. This poses a risk to sensitive biological areas that have been identified (Wassenaar et al., 2007). In Mexico, livestock in pastures with introduced grasses has been the principal cause of tropical dry forest conversion (Trilleras et al., 2015).

2.2.1.3 Fish (wild, marine, and freshwater fisheries and aquaculture)

Wetlands, rivers, lakes, estuaries, and oceans have long been vital sources of fish and shellfish products, and their contributions are widely recognized as one of the healthiest sources of animal protein for human consumption (Nesheim & Yaktine, 2007; Fernandes et al., 2012; FAO, 2014b). This NCP, however, is compromised in some locations and species by the contamination of fish tissues with toxic compounds (United Nations Oceans and Law of the Sea, 2016; Bonito et al., 2016). The relative contributions of fish to humans are indicated by yield, consumption, and economic data based on job number and economic benefits (see Box 2.1 and 2.2). In the Americas, wild capture fisheries produced 17.9 million tons in 2012, and aquaculture yielded about 3.2 million tons, or about 15% of the total (FAO, 2014b; United Nations Oceans and Law of the Sea, 2016). Most of the wild capture fishery yield is marine, and inland continental waters (mostly freshwater) produce only about 3% of the total capture in the Americas, which notwithstanding the low absolute value can be locally important (FAO, 2014b). The locations of exceptionally important freshwater fisheres in the Aemricas include the Amazon River in South America (e.g. Bastos & Petrere, 2010; Isaac et al., 2015) and the Great Lakes and Upper Mississippi River in North America (GLMRIST, 2012). While the ecological production of wild fisheries is a natural service, aquacultural production is largely a function of human efforts. Aquaculture in the Americas (mainly Chile, Brazil and the USA) contributes nearly 5% to the world fish yield (FAO 2014b), or at a subregional level is constitutes 9%

to the total fish/shellfish yield in North America, 13% in Mesoamerica and the Caribbean, and 22% in South America (FAO, 2014b). Aquaculture production often comes at environmental costs.

Fish is the major source of high quality protein in many countries (e.g. Islam & Berkes, 2016; Hanazaki et al., 2013), but is less than 6% of the protein in the average diet in the Americas (FAO, 2014b). Overall, North American per capita consumption (~20 kg/yr) is greater than in Mesoamerica, the Caribbean and South America (~10 kg/yr) (FAO, 2014b). Variability in consumption among nations is high and may be related to the proximity of a population to marine ecosystems, as well as cultural practices and preferences. At the extremes, fish consumption averages only 2.2 kg/yr in inland Bolivia and 53.4 kg/yr on the Caribbean island of Antiqua (FAO, 2014b). Commercial fisheries are also major sources of animal feed, fertilizer, and fish oil (FAO, 2014b). Other products include glue, pearls, buttons, and medications.

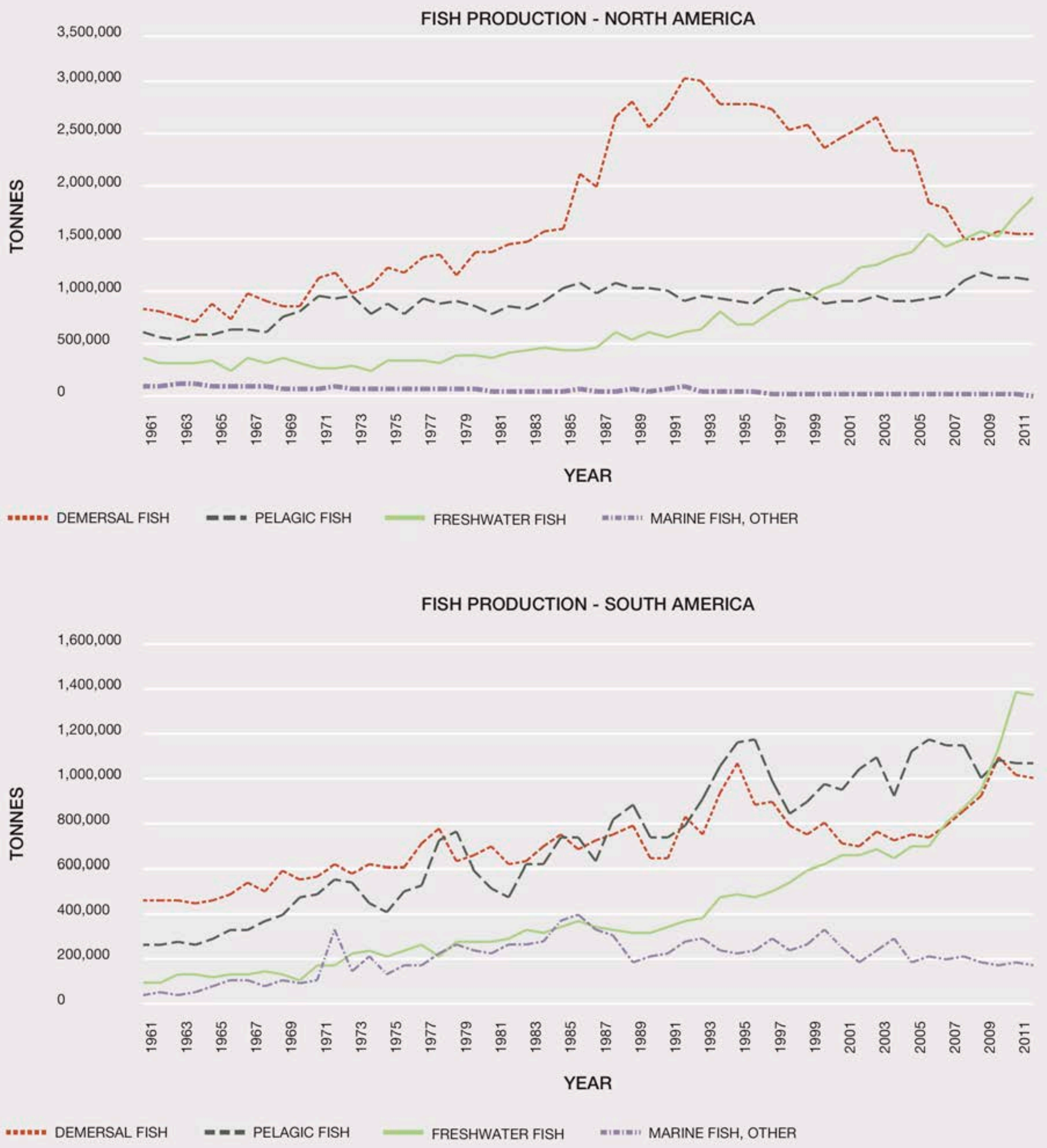
Commercial fisheries provide employment for about 325,000 people in North America and 2,444,000 people in Mesoamerica, the Caribbean and South America (FAO, 2016d). Job numbers and per capita income vary widely, depending on the specific fishery and its location. While fishers comprise less than 1% of the North American work force, in the Caribbean they constitute about 5% (Masters, 2014). Employment is, however, often physically difficult and hazardous; the second most deadly job in the USA (USBC, 2016). Sport fisheries also provide jobs for many people. In the USA alone in 2010, they supported over 820,000 jobs and $35 billion in salaries and wages (FWS, 2011). Aquaculture employs 356,000 people in Mesoamerica, the Caribbean and South America and 9,000 in North America (FAO, 2016). Small scale and subsistence fisheries play a major role in providing food and income security for rural and coastal communities, particularly in Mesoamerica and South America (e.g. Hanazaki et al., 2013), but also among the indigenous group of North America (e.g. Islam & Berkes, 2016). Weeratunge et al. (2014) emphasized the contribution of the material, relational and subjective dimensions of small-scale fisheries to the well-being of individuals and communities. The role of women directly or indirectly involved in many fisheries contributes to household security throughout the Americas. For instance, the Sirionó of Bolivia, fishing supplies an important contribution to family nutrition (23%), one that is accessible to women and children who practice the activity daily (Townsend, 1995).

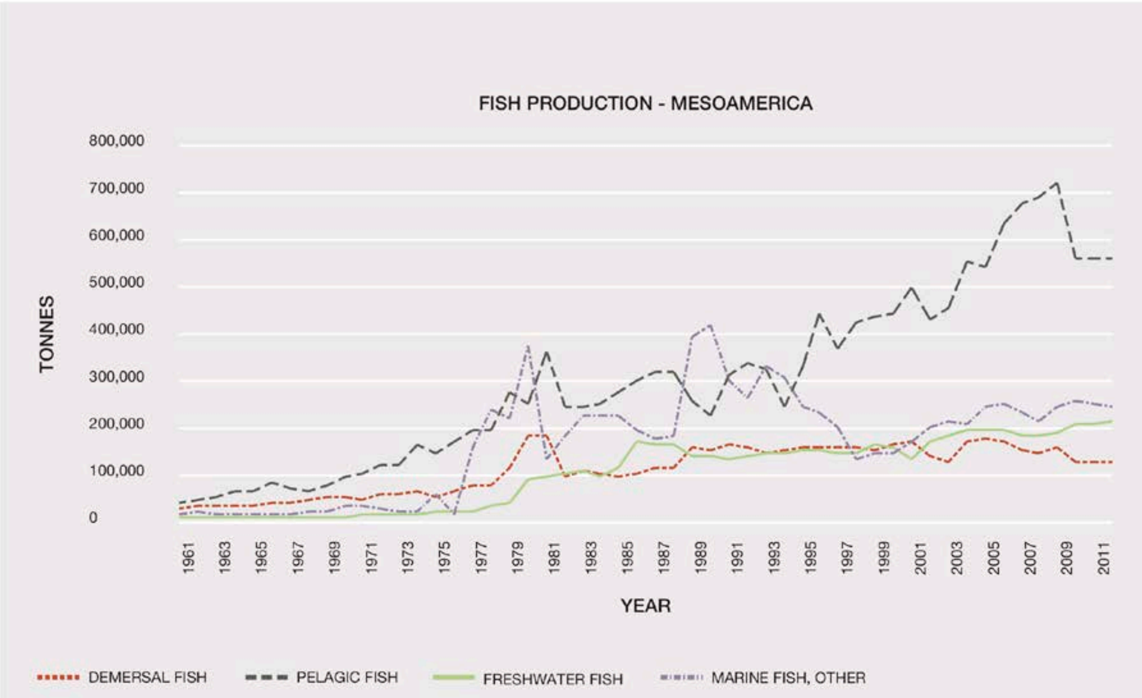
The world's commercial production of wild fisheries increased until about 1990 and then plateaued (FAO, 2014b), largely in response to reaching nature's sustainability limits (Pauly, 2002). While the wild fish yield has been stable since the 1990s, catch composition has changed, as some stocks were depleted and others increased in importance (Pauly, 2002; Rose & Rowe, 2015; Pershing et al., 2015). In the Americas, the yield of wild fisheries also peaked in the 1990s (Figure 2.5) and has declined somewhat in North America and in the Caribbean (FAO, 2014b), where overfishing threatens up to 70% of coral reef ecosystems (Burke et al., 2011). The steep increase of freshwater yield in North and South America, shown in Figure 2.5, parallels the increased importance of aquaculture as wild capture fisheries plateaued. Aquacultural yield has increased rapidly since 1990, allowing the upward trend in total fish yield to be maintained (FAO, 2014b).

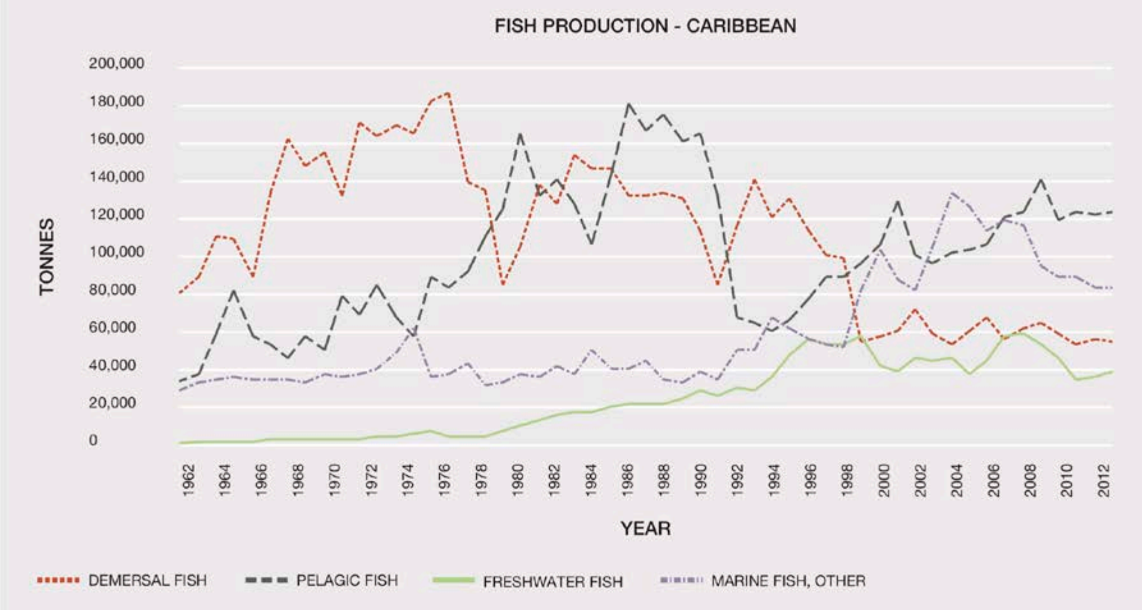
After sustained increases between 2004 and 2011, the fraction of fisheries catch certified for its legal origin and process by the Marine Stewardship Council has recently plateaued in the Americas at less than 10%, which is a value similar to East Asia and the Pacific, but less than half the percentage attained in Europe (Figure 2.6).

Between 2000 and 2014 in North America, the number of people employed declined by 7% in commercial fisheries and 44% in aquaculture (FAO, 2016d). During the same period in the rest of the Americas, the number increased by 37% in commercial fisheries and 66% in aquaculture. These different trends may be related to competition and the physical difficulty and dangers associated with wild fisheries in North America, leading to further mechanization and job replacement; while in the other three subregions, a lack of jobs and unregulated fisheries may drive many people to this sector to increase local food and livelihood security.

Figure 2 5 **Fish production (tons) in the Americas per subregion, 1960s–2012. Source: FAO (2017). FAOSTAT Statistics Database. http://www.fao.org/faostat/en/#data/CL. Data accessed: March 19, 2017.**

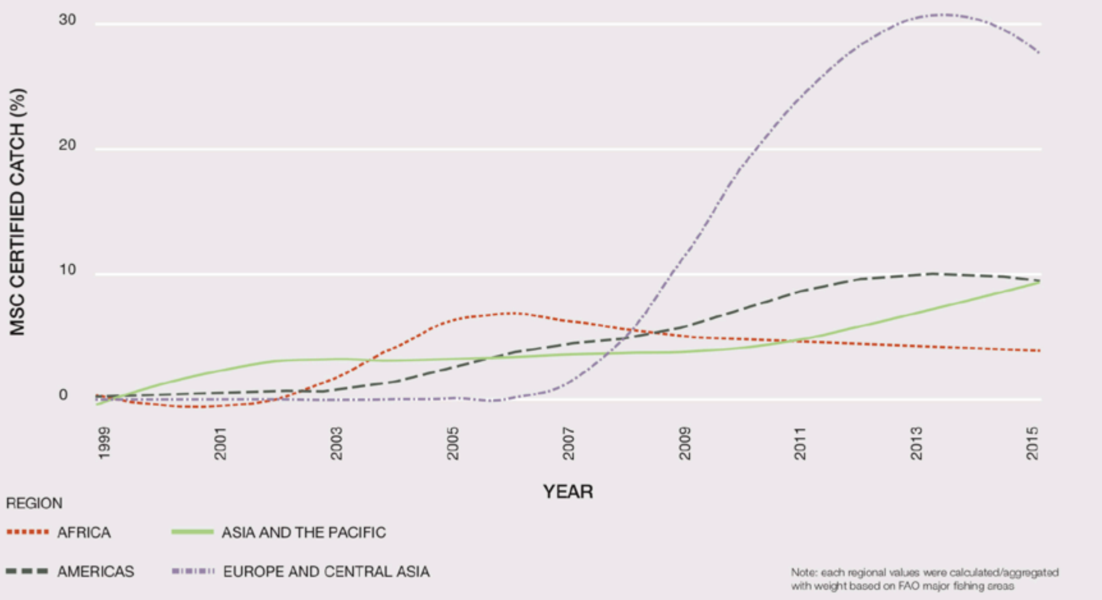






The main drivers of future impacts on the provision of these services are increased demand for fish, which is a function of population number and per capita consumption. The pressure on wild fisheries is moderated by harvest regulations, improved fisheries techniques, and aquacultural development (FAO, 2016d.) Total aquacultural yield is projected to increase significantly in the future while the total yield from wild fisheries remains generally stable as composition changes (FAO, 2014b). Fish protein consumption depends largely on availability and price changes. The economic setbacks of some countries in the Americas has pushed more people into lower-income fisheries (mainly artisanal, small-scale fisheries), which often exploit near shore stocks unsustainably. Future trends indicate that the overall number of jobs will decrease in response to mechanization.

Figure 2 6 Percentage of fisheries catches per region certified by the Marine Stewardship Council. Indicator data source: Marine Stewardship Council. The figure prepared by Task Group on Indicators and Knowledge and Data Technical Support Unit.



Box 2.1 Economic value of fisheries contributions to human quality of life.

Measuring the economic value of fishery services to human quality of life in a manner that allows comparisons across different ecosystem types and subregions is complicated by inadequate data. Most comparable economic estimates of fisheries' ecosystem services, based on annual per hectare monetary values, are for wetland ecosystems with readily definable boundaries (all values reported here are adjusted to 2016 USA dollars). They vary widely among fishery locations and conditions. For inland wetlands, Woodward and Wui (2001) estimated benefits between $488/ha/yr and $25,394/ha/yr for numerous sites in North America and Europe, and Seidl and Moraes (2000) estimated $86/ha/yr for the Pantanal in Brazil. Early estimates for coastal wetlands include $133 (Costanza & Farber, 1987), $179 for shrimp alone (Barbier & Strand, 1998), and more recently, $3,959 for combined fishing and hunting (Camacho-Valdez et al., 2013).

The dock-side value of marine and lake catch provides a high estimate of natural service benefits. However, outside the USA, where it was recently valued at $5.5 billion per year (NMFS, 2016), the data for dock-side sales (points of first sale) are inconsistently documented. A rough estimate of the world dock-side value per unit area of oceans and the Laurentian Great Lakes in North America was estimated, using production data from FAO (2014 a, b) and the Great Lakes and Mississippi River Basin geographical area (GLMRIST, 2012) and an assumed $1/kg dockside value, like that of the USA. These estimates found a much lower economic value for oceans and the Great Lakes, compared to wetlands, with $0.025/hectare for oceanic fishery services and $0.035/hectare for the Laurentian Great Lakes.

Box 2.2 Caribbean coral reef contribution to fisheries and human quality of life.

Coral reefs in the Caribbean provide a wide range of services for almost 40 million people, which affect livelihood, economic progress, food security, cultural expressions and communion with nature (Jackson et al., 2014). They are the basis of the tourism and fishing industries in the insular Caribbean and most of Mesoamerica and the southeastern USA (UNEP, 2010). Both tourism and fisheries development are major contributors to GDP and employment in the region. It is estimated that nearly 350,000 persons were employed in the fishing section in 2011 in 17 Caribbean countries including Guyana and Surinam; this represents about 5% of the total work force (Masters, 2014).

The annual value of services provided by Caribbean coral reefs has been estimated at between $3.1 billion and $4.6 billion, and the total economic impact of coral reef-associated fisheries was about $0.8–1.1 million per year in Tobago and $0.5–0.8 million per year in St. Lucia (Burke and Maidens 2004). Mahon et al. (2007), showed that as the fish moved through the various market pathways to the consumer it increases in value, contributes to livelihoods, and that the overall additional value was 2.6 times the landed value of the fishery. In 2011-2012, at ex-vessel prices (the point of first sale) the value of the marine capture fishery production for the Caribbean region was estimated at $392.9 million annually and the aquaculture fishery at $28.9 million annually, giving a total value of approximately $421.8 million over the period (Masters 2014). It is estimated that the continued decline of coral reefs could cost the region between $350 million and $870 million per year by 2050 (Burke & Maidens, 2004; Agard & Cropper, 2007).

2.2.1.4 Wildlife

Wild game provides an important food resource to many people of the Americas, especially indigenous peoples and local communities, but it also has important recreational and cultural values (sections 2.2.5, 2.2.6, 2.5.1). In Bolivia, for instance, if indigenous people had to replace the protein contributed by nature through their hunting efforts, they would need to pay from $60 to $120 per family per month (Copa & Townsend, 2004; Townsend & Gomez, 2010), and the estimated monetary value of this NCP in the state of Santa Cruz alone is between $3 to $24 million a year (Gobierno Autonomo Departamental de Santa Cruz*,* 2009). In Caribbean island nations, which import most of their food, especially meat, wildlife management can also be a way to search for food sovereignty (e.g. captive breeding programs for some Neotropical mammals (Singh et al., 2016). Meanwhile, in Mesoamerica many of the harvested wildlife species are those whose adaptation to humans (Linares, 1976). For example, the Maya consciously use their *milpa*, or garden plots, to attract game and increase their hunting potential (Jorgenson, 1993; Santos-Fita et al., 2012). Today's Mesoamerican indigenous groups are mostly sedentary, without access to extensive hunting territories and rely principally on their agricultural production (Santos–Fita et al., 2012), but they still maintain an important cultural and spiritual relationship with wildlife, even though it has become mainly a supplement to their family's nutrition (Garcia del Valle et al., 2015; Santos Fita et al., 2015). Finally, Table 2.3 presents summary information about ungulates that are key subsistence species in North America.

Table 2.3. Ungulate species most utilized in North America.

Scientific name	Common name	Distribution	Group size - Land use	Source
Rangifer tarandus	Caribou	Large populations in Arctic, subarctic and boreal regions of Canada, Alaska	Large herds - Migratory	White (1975)
Alces alces	Moose	Boreal regions of North America	Resident, and some migratory	Franzmann & Schwartz (2007)
Cervus elpahus	Elk	Western North America - once the most widespread North American deer ranging almost coast to coast, now found primarily in western mountain regions	Small groups, local seasonal migration	Thomas & Toweill (1982), Houston (1982)
Antilocapra americana	Pronghorn antelope	Dry open areas, including brushlands, grasslands, and deserts of interior western and central North America. In Canada, pronghorn occur only in southern Alberta and Saskatchewan	Small to larger groupings, generally restricted due to limited habitat	O'Gara & Yoakum (2004)
Bison bison	Bison, Buffalo	Wood Bison subspecies- boreal forest in the Yukon and Northwestern Canada. Plains Bison - Southern Great Plains of North America.	Originally large herds- now small groupings or local herds restricted in movement	Lott (2002)
Oreamnos americanus	Mountain goat	Mountainous regions of western North America	Small groups, local movements	Festa-Bianchet (2008)
Ovis canadensis, Ovis dalli	Rocky mountain bighorn sheep, Dall's mountain sheep	Southern British Columbia and southwestern Alberta, Canada to northwestern USA, including Alaska	Social animals with local altitudinal migration	Valdez & Krausman (1999)
Odocoileus hemionus/ Odocoileus virginianus	Mule deer/ White-tailed deer	From Mexico to Alaska/ North America through northern South America	Individuals or small groups, local, possible altitudinal migration	Halls (1984), Wallmo (1981) Wilson & Ruff (1999)
Ovibos moschatus	Muskox	Islands and mainland in the Canadian Arctic and Greenland, introduced in parts of Alaska	Localized groups	Wilson & Ruff (1999)
Source: Kuhnlein & Humphries (2017).				

A great diversity of species are harvested within susbsitence (non-commerical) economies in the Americas. Indigenous groups incorporate a great diversity of wildlife into their diet and consume at least 527 animal species of freshwater, marine, and terrestrial organisms (Kuhnlein and Humphries 2017). Ungulates are the most consistently hunted wildlife group used for food and subsistence (Robinson & Redford, 1991; Townsend & Rumiz, 2004; Iwamura et al., 2014; Townsend & Gomez, 2010; Constantino, 2016), except where their use might be prohibited by cultural controls such as cultural preferences (Ayala, 1997), taboos (Reichel–Dolmatoff, 1971; Baleé, 1985, 1993), the absence of a species' preferred habitat and/or its deterioration (Cuellar, 1997), or over-exploitation (Mittermeier, 1991; Peres, 1990, 1991; Atunes et al., 2016). Indeed, some tribes have strict taboos which dictate which taxa are edible. For instance, the Ayoreo tribe of Bolivia and Paraguay forbid hunting most mammals and focus on land tortoises (Ayala, 1997), or the Kalapalo people of Brazil consider that all terrestrial mammals are taboo, but can consume primates (Basso, 1973). Urbani (2005) reviewed 56 wildlife hunting publications in South America, finding that 33 of the studied human groups included primates in the species they hunted. Among mammals, hoofed animals are very often top on the list of species used in all the Americas, but waterfowl and game birds are also important, depending on specific ecosystems.

It has been estimated that sustainable hunting in tropical forests requires at least 1 km^2/person (Robinson & Bennet, 2000). In this context, sustainable production of the 8 most-harvested species in Mesoamerica and South America (i.e. collared peccary, *Tayassu tajacu*; red brocket, *Mazama americana*; grey (or brown) brocket, *Mazama gouazoubira*; South American tapir, *Tapirus terrestris*; lowland paca, *Cunniculus paca*; brown agouti, *Dasyprocta variegate*; nine-banded armadillo, *Dasypus novemcinctus*; and Southern America coati, *Nasua nasua*) could reach about 1.4 kg/ha/yr of wild meat in natural tropical forests (Gobierno Autonomo Departamental de Santa Cruz, 2009). However, sustainable production is completely contingent on maintaining the wildlife production lands in good condition (Altrichter, 2006; Silvius et al., 2004; Townsend, 2010; Alvarez & Shany, 2012).

In North America, wildlife hunting requires a permit, so species populations can be managed and harvest levels controlled. Indigenous people have prioritized access in some areas, including Canada where rights to harvest wildlife and fish are protected where treaties have been signed. Historically, wildlife harvesting represented a significant proportion of protein consumed; however, decreases in biodiversity of wildlife species, habitat degradation and decreased access (e.g. physical and regulatory) has contributed to a steep decline in wildlife harvesting in many areas of the Americas. While only 6% of the population participates in recreational hunting (Mahoney, 2009), these programs generate revenue, not only to government agencies via the permit process, but also an estimated $25 billion in retail sales yearly and $17 million in wages and salaries is generated yearly (IAFWA, 2002). The tax revenue to the USA from retail and permits is estimated to be more than $2.4 billion per year and trends in participation of recreational wildlife use in USA have been stable over the past few decades. In addition, the sale of hunting permits provides in large conservation benefits. In Canada, for example, the revenue generated from the sale of habitat conservation stamps, affixed to the migratory game birds permits, funds habitat conservation projects, and since 1985, >$50 million has been generated to support 1,500 habitat conservation projects across the country. Mexico uses wildlife management units as a strategy to combine conservation of game species with economic activities via the sale of animals or hunting. These wildlife management units can be part of community activities or take place on private lands (http://www.biodiversidad.gob.mx/usos/UMAs.html). In some instances, wildlife on common lands are managed as a common use resource, whereby different organizations can develop hunting activities and sometimes there is no payment for hunting by external people, but rather an exchange for external merchandise. These wildlife management units have been underway in Mexico since 1997, and currently 37% of Mexican municipalities have them, recording 417 species (http://www.biodiversidad.gob.mx/ usos/UMAs.html).

2.2.1.5 Organic products

Over the past two decades, in the face of increase used of pesticides on plants and antibiotics and growth hormones in animal products, more consumers are buying organic food to assure their quality of life. They are willing to pay a premium for better health, environment quality and animal welfare (Dimitri & Greene, 2002). For example, the market value for organic foods in the USA, especially fruit and vegetables, nearly tripled over the last decade (Figure 2.7), and the area of land under organic farming has increased over the past decades (Figure 2.8, section 2.2.8).

According to FAO (2007), small-scale farmers have been successful in adopting organic practices and marketing their products (e.g. in supermarkets or farmers markets in cities where organic and local vegetables are sold). In this study, covering 14 farmer groups with more than 5,100 small farmers, each with about two hectares of land in six countries (Argentina, Costa Rica, Dominican Republic, El Salvador, Guatemala and Mexico), organic farming systems were found to 1) embody many elements of sustainability that make them effective tools to help reduce poverty and improve food security including; 2) support long-term commitments to soil fertility, particularly reducing soil erosion and degradation or desertification; 3) reduce external energy consumption and water requirements; 4) enhance the value of in knowledge-intensive rather than capital- and resource-intensive practices; 5) link traditional knowledge with modern methods such as bio-controls and efficient nutrient management; and 6) integrate traditional knowledge, joint problem solving and farmer-to-farmer exchange to improve community relations and lead to greater involvement and commitment of producers.

Figure 2 7 USA organic food sales by category. Source: USDA (2014).

https://www.ers.usda.gov/topics/natural-resources-environment/organic-agriculture/organic-market-overview.aspx Date accessed: April 4, 2017.

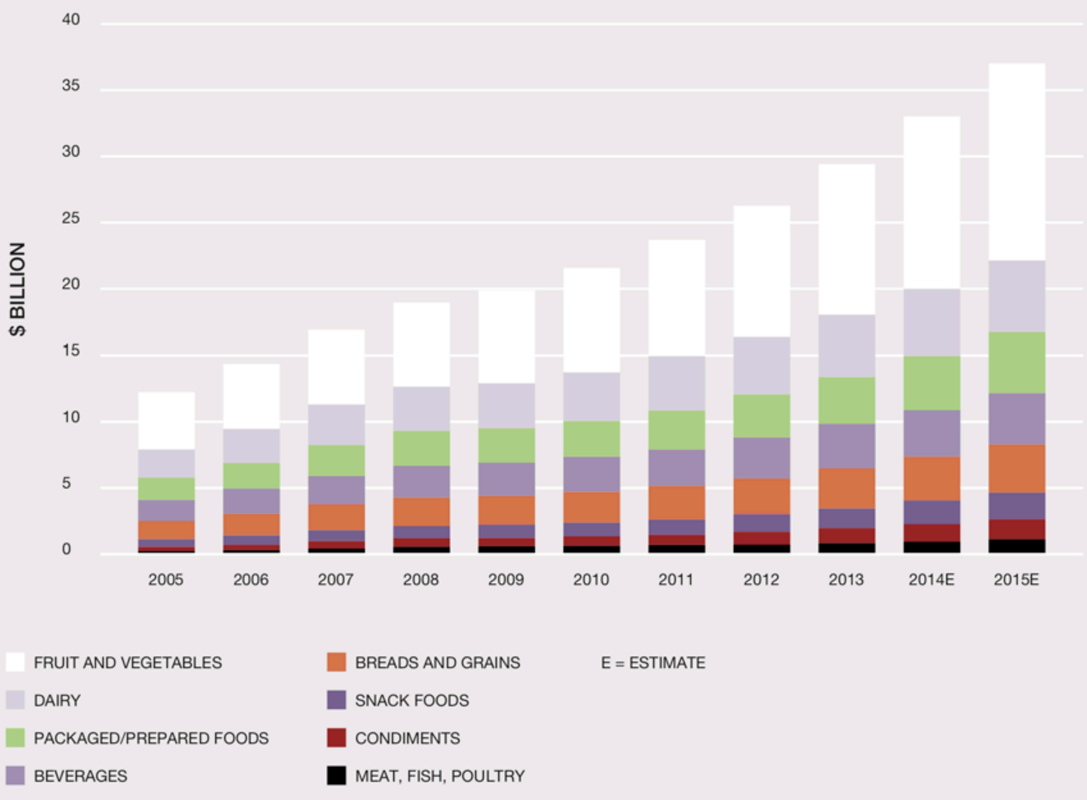
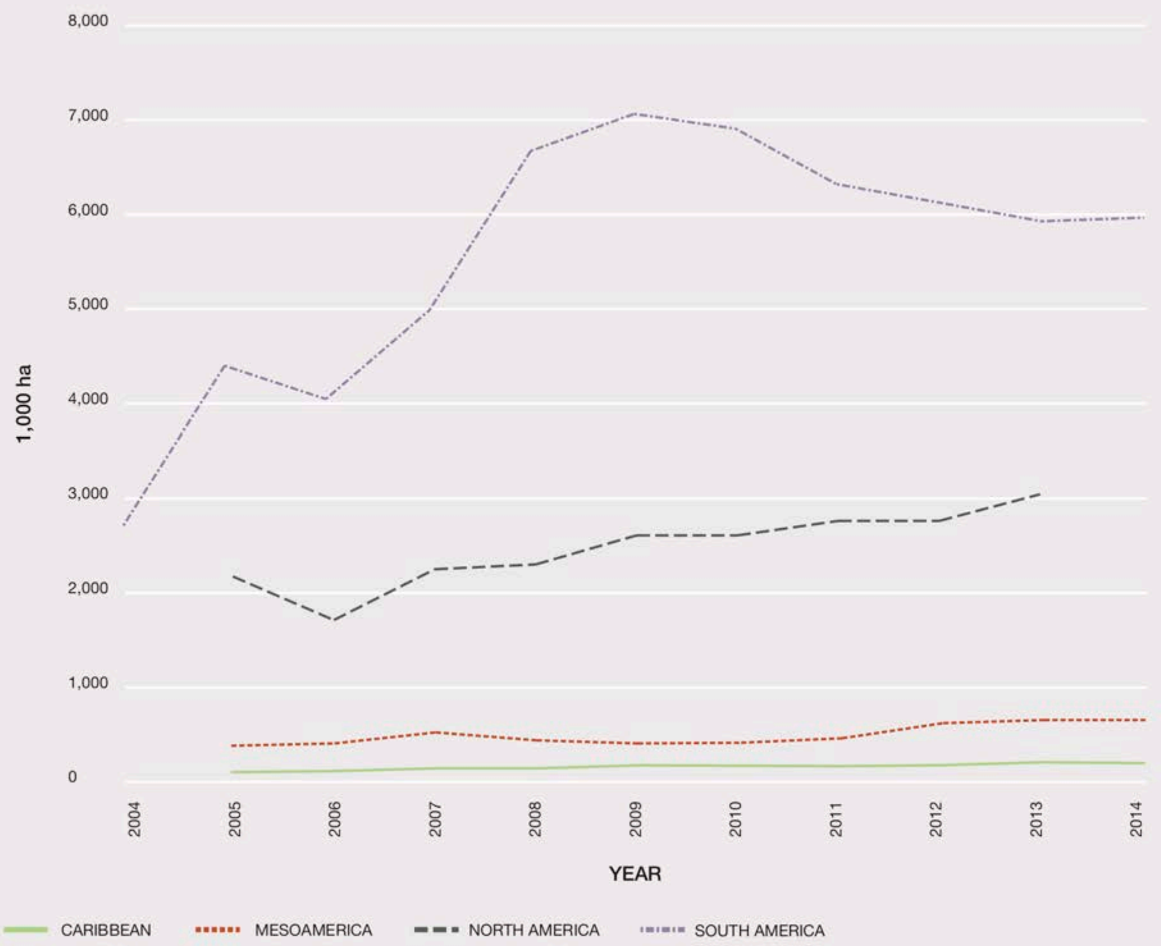


Figure 2 8 Trends in agricultural organic area per subregion.
Source: FAO (2017). FAOSTAT Statistics Database.
http://www.fao.org/faostat/en/#dataRL. Date accessed: November 9, 2017.



2.2.2 Materials and assistance

Timber and fiber are essential provisioning services for a good quality of life. They provide shelter through construction materials, clothing, and raw materials for industries and manufacturing. Extraction, processing, production and trade of these services are also important livelihood activities of many individuals worldwide (section 2.3.5). Production rates of this NCP have increased considerably over the last several decades, helping improve the quality of life for many with some associated negative social and environmental impacts notwithstanding. However, rates of production have slowed down and are expected to continue declining as new technologies and production substitutes emerge. There are stark variations between subregions in production and consumption of various timber and fiber services, as shown below.

2.2.2.1 Timber

North America is the largest producer and, in some cases, consumer of timber products. In this subregion, for instance, coniferous sawnwood greatly outpaces other subregion’s production, peaking in the late 1990’s and early 2000’s (Figure 2.9).

Figure 2 9 Production, imports and exports of sawnwood (coniferous and non-coniferous) by subregion. Source: FAO (2017). FAOSTAT Statistics Database. http://www.fao.org/faostat/en/#data/FO. Date accessed: February 6, 2017.

Note: The stat_smooth function was applied in R (ggplot2 package) to get the smooth lines.

Countries with the highest wood removals in the Americas are the USA, Brazil and Canada, as partially reflected in their gross value-added USA dollars in the forestry sector (Table 2.4). In 2011, approximately 858 million m^3 of wood were removed in the Americas region alone, and between 1990 and 2011, annual wood removals in North America were varied, with a decrease following the 2008 financial downturn. Furthermore, the share of woodfuel also varies by subregion, accounting for only 9% of total removals in North America, whereas in South America and Mesoamerica it accounts for 78% and 88%, respectively (Figure 2.10).

Figure 2 10 Annual wood removals in the Americas by subregion from 1990 to 2014. Indicator data source: FAO. The figure prepared by Task Group on Indicators and Knowledge and Data Technical Support Unit.

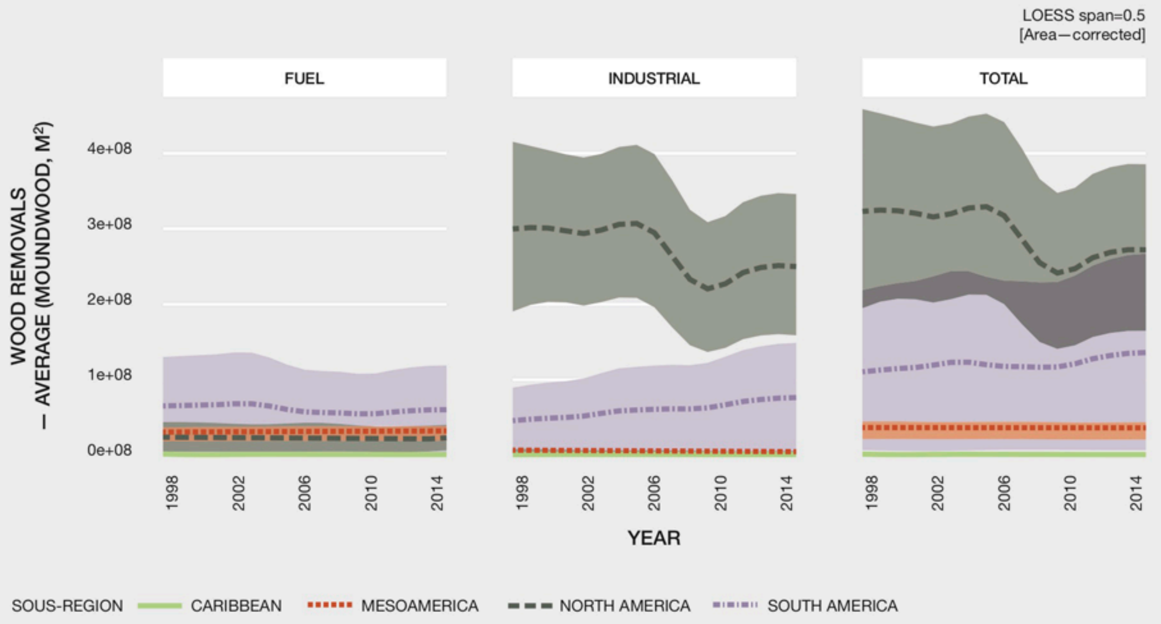


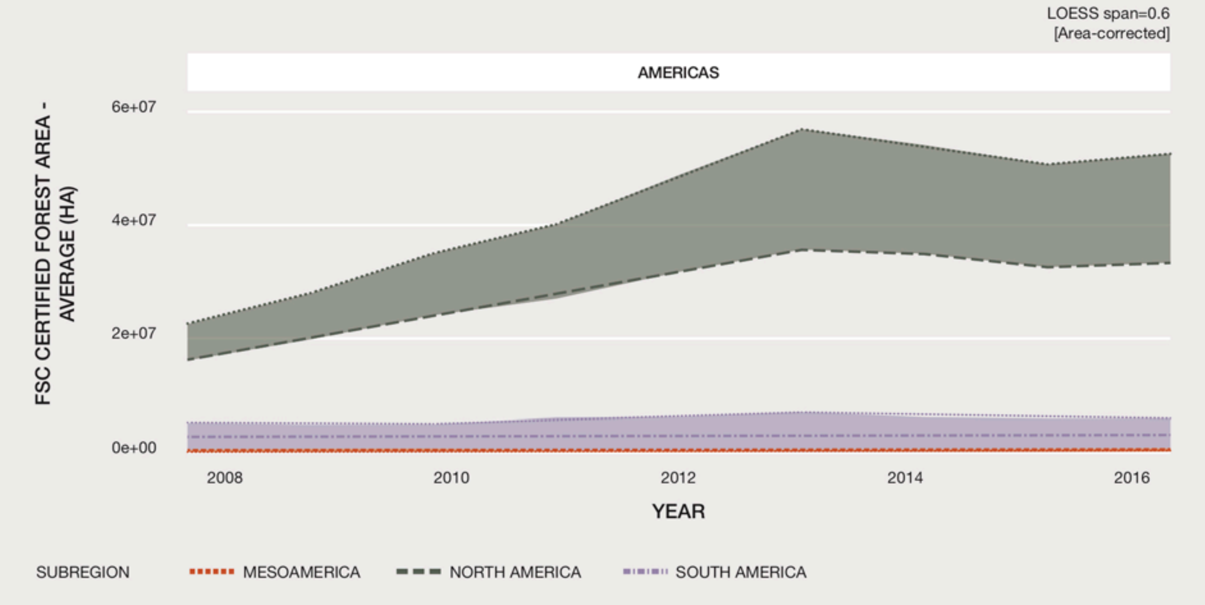
Table 2.4. Top ten countries value-added (US $ in millions in 2011 prices and exchange rate) in the forestry sector 1990-2011.

Country	1990	1995	2000	2011	AAGR
USA	110,346	132,476	135,498	95,664	-0.7
Brazil	24,732	24,522	19,928	22,513	-0.4
Canada	26,392	41,116	43,339	19,789	-1.4
Chile	2,605	4,449	5,432	7,596	5.2
Mexico	7,123	5,618	7,021	6,954	-0.1
Argentina	1,607	1,19	1,477	2,055	1.2
Colombia	2,192	1,906	1,956	1,826	-0.9
Ecuador	1,803	2,421	1,946	1,741	-0.2
Venezuela	658	747	675	1,43	-25.3
Peru	542	702	849	1,316	-24.9
AAGR: Average Annual Growth Rate. Source: Forest Resources Assessment (2015).					

Timber extraction, as with many other production activities, is driven by various underlying factors interacting synergistically in space and time (Geist et al., 2006). For instance, cultural factors drive preferences for wood products; in many cases they covary with human population growth, technological factors, and industrial growth. These drivers tend to be regional to global in scope, act in complex ways and are usually mediated by institutional factors (Bryan, 2013; Lambin et al., 2001). Cultural preferences for sustainably harvested wood have continued to drive market-based certification schemes in the forestry sector (MacDicken et al., 2015). Regionally, since 2000, North American timber operations top other Americas subregions in the total area under international forest certification schemes, as in the case of the Forest Stewardship Council certification (Figure 2.11).

Figure 2 11 **Temporal trend in the hectares certified by the Forest Stewardship Council (FSC) in the Americas per subregion since 1998.**

Indicator data source: Forest Stewardship Council. The figure prepared by Task Group on Indicators and Knowledge and Data Technical Support Unit.



Meanwhile, technological advancements continue to play an increasingly important role in driving production of forest products. Remote sensing technologies, for instance, facilitate and inform forest product operations, policies and decision-making (Romijn et al., 2015). Also, the increasingly widespread use of electronic media and mobile technologies has substantially reduced demand for paper products in many parts of the world, as have improvements in the production and commercialization of wood substitutes (FAO, 2016a). Income growth continues to dictate both demand and production of wood products. The largest economies in the world, particularly the USA, China, Germany and the United Kingdom, lead as major consumers of many forest products, including industrial roundwood, wood pellets, sawnwood, and paper and paperboard (FAO, 2016a), showing a strong link between demand and economic might. Emerging markets and production sites in China and India have been pivotal in driving timber demand and production in the Americas and elsewhere the last few decades, owing to their robust manufacturing sectors as well as the expansion of their economic middle class. Policy and institutional factors also have determined wood product demand. International agreements and policies in Europe, for instance, have spiked demand for wood biofuels, as have forest policy incentives for some products in certain locales (e.g. Farley & Costanza, 2010; Lawler et al., 2014). Forest management institutions and local governance systems are key mediators between demand forces and production trajectories in the forestry sector (FAO, 2016b).

Forest and timber extraction activities contribute to biodiversity loss through fragmentation, habitat destruction and single-species plantation systems (Lawler et al., 2014) (see also Chapter 4). Some NCP are negatively affected through soil degradation, reduced water regulation and quality, as well as impeded carbon storage capacities. Further, timber activities may lead to losses of cultural traditions and diversity, and reduced access to key ecosystem services for traditional forest-dependent communities. However, positive ecological effects may ensue through restoration practices such as reforestation or afforestation activities in previously degraded/cleared lands (FAO, 2016b). Some positive social impacts include employment opportunities and subsistence means for rural populations, overall economic growth, provision of energy supplies, and building materials (FAO, 2016b; Whiteman et al., 2015).

2.2.2.2 Fibre

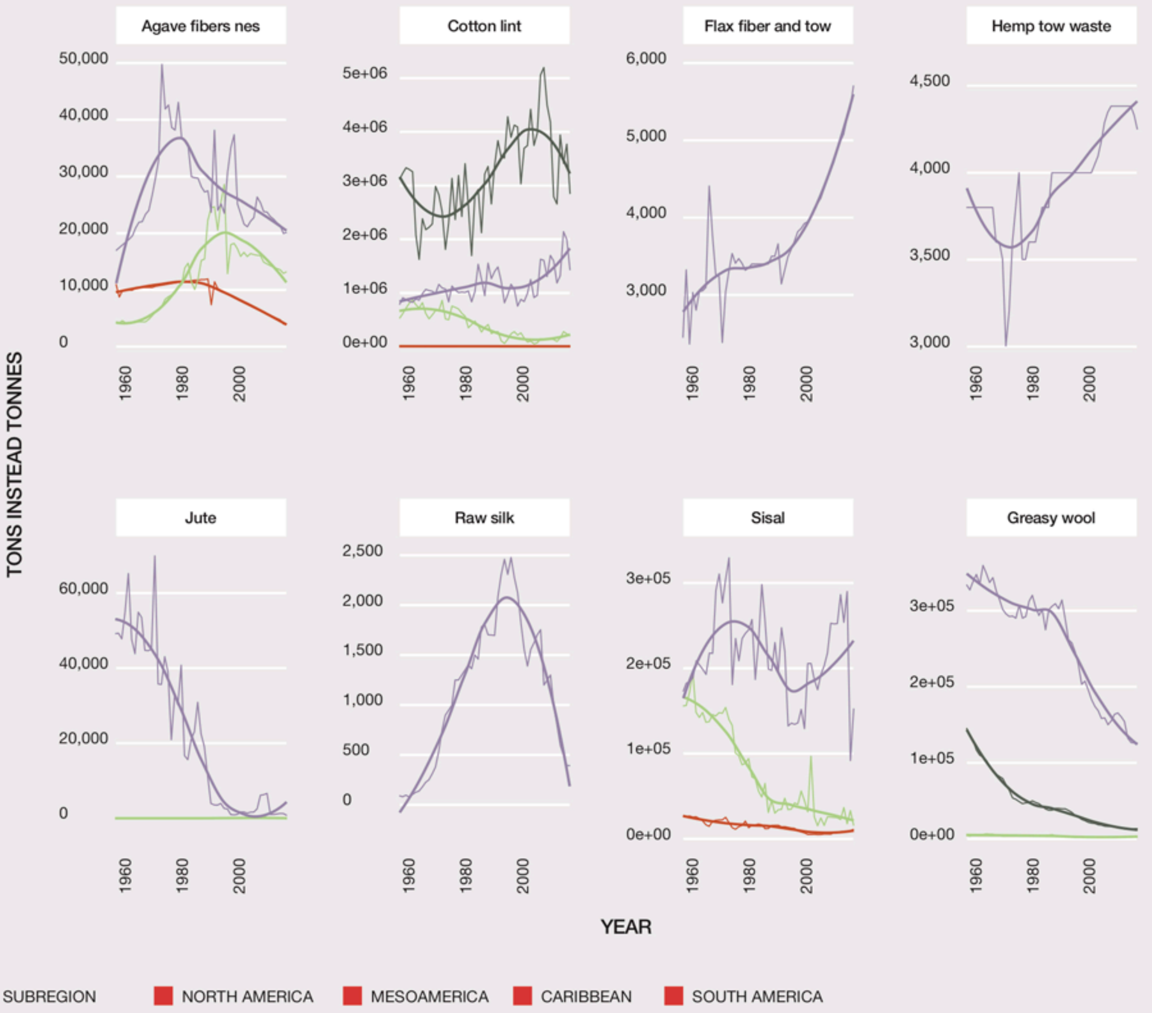
Fibres have been used by humans since early times, and are key components of well-being through provision of shelter, clothing, and other benefits. They are used to fabricate products such as building materials, paper, cordage, textiles, baskets, brooms, and rugs. Aside from plants, fibers are also obtained from animal and mineral sources. Fibres have been widely used in the Americas for millennia. Cotton, flax, hemp, jute and sisal are the most commonly produced vegetable fibres in this region (Table 2.5). North America stands out as the highest producer of cotton, while production in South America is increasing (Figure 2.12).

Table 2.5. Important plant fibers in the Americas and their uses.

Plant	Scientific name	Family	Description	Diameter	Use
Cotton	*Gossypium hirsutum; Gossypium barbadense*	Malvaceae	The world's most popular natural fibre, cotton is almost pure cellulose, absorbs moisture easily.	Fibre length varies from 10 to 65 mm, and diameter from 11 to 22 microns	Cotton cloth
Flax	*Linum usitatissimum*	Linaceae	The fibre is a cellulose polymer, its structure is stronger and stiffer and absorb and release water quickly.	Flax fibres range in length up to 90 cm, and average 12 to 16 microns in diameter	Linen
Hemp	*Cannabis sativa*	Cannabaceae	Around 70% cellulose, containing low levels of lignin (8-10%). Is a heat conductor, resist mildew and has natural anti-bacterial properties.	The fibre diameter ranges from 15 to 50 microns.	Hemp cloth, canvas, cordage
Jute	*Corchorus spp.*	Malvaceae	Jute is long, soft and shiny with high insulating and anti-static properties, moderate moisture regain and low thermal conductivity.	The fibre length ranges from 1 to 4 m and a diameter of from 17 to 20 microns	Burlap
Sisal	*Agave sisalana; Agave fourcroydes*	Agavaceae	Sisal is coarse, hard, strong, durable and stretchable. Resists the saltwater deterioration, has a fine surface texture appropriate for a wide range of dyes.	The fibre measures up to 1 m in length, with a diameter of 200 to 400 microns.	Cordage, matting
Source: adapted from Levetin & McMahon (2008), FAO (2009) International Year of Natural Fibres (http://www.naturalfibres2009.org/en/fibres/index.html)					

Figure 2 12 Production of vegetal and animal fibers in the Americas, 1961–2013.

Note: Fibers from ginning seed cotton that have not been carded or combed; agave fibers include inter alia: Haiti hemp (*Agave foetida*); henequen (*A. fourcroydes*); ixtle, tampico (*A. lecheguilla*); maguey (*A. cantala*); pita (*A. americana*); Salvador hemp (*A. letonae*). The leaves of some agave varieties are used for the production of alcoholic beverages, such as aquamiel, mezcal, pulque and tequila; Sisal (*Agave sisilana*) is obtained from the leaves of the plant. It also is used as an ornamental plant; the production of jute includes white jute (*Corchorus capsularis*); red jute, tossa (*C. olitorius*). The stat_smooth function was applied in R (ggplot2 package) to get the smooth lines. Source: FAO (2017). FAOSTAT Statistics Database. http://www.fao.org/faostat/en/#data/QC. Date accessed: April 10, 2017.



South America and Mesoamerica have been important producers of plant fibres, such as agave and flax, albeit with decreasing trends recently (Figure 2.12). Production of these fibres is strongly characterized by peaks driven by diverse underlying factors. For instance, since the 1960s, agave production has been intermittent with sharp increases starting in the 1970’s in South America and in the 1990’s for both Mesoamerica and South America. The Caribbean, on the other hand, has shown a relatively stable trend towards decline since the 1960s for agave. Production of sisal has shown a similar behavior, although with more abrupt declines recently for some subregions. Production of jute in South America has shown similar peaks, but with a more prominent decreasing tendency since the 1960’s (Figure 2.12).

Production of certain animal fibres also shows sharp production peaks. In South America, raw silk production, for instance, is declining after a peak in the mid-1990’s (Figure 2.12). Wool production has also declined over the last decades, particularly in South America. This is in large part due to the increasing use of synthetic substitutes for clothing (Figure 2.12).

As with timber products, cultural factors play a pivotal role in driving large-scale fiber production and demand (Graham-Rowe, 2011). Many consumers have preferences for particular types of plant and animal fibers, such as skins, furs, wood-based fibers, cotton, silk, wools and hairs used to fabricate a gamut of product types including clothing, fashion accessories, ornaments, and furnishings. These preferences, in turn, are driven largely by fashion trends propagated through globalized media. Population growth also constitutes a significant driver of fiber production and consumption. In some cases, demand for certain types of animal and plant fibers has stagnated or decreased thanks to the more pervasive use of alternative synthetic. Agricultural biotechnologies also continue to strongly influence fiber production (Ali & Abdulai, 2010), as do policies and institutional factors largely through regulatory mechanisms such as controls and restrictions on trade, poaching and illegal harvesting of fibers.

The environmental impacts associated with fiber production depend on the type of fiber, the extraction methods, as well as the scale of production (Clay, 2004). This includes impacts through substantial pesticide use, soil degradation and salinization, and water diversion for irrigation. Other environmental impacts include significant reductions in the populations of wild species used for vegetable and animal fibers that may lead to vulnerability of population declines for those species. Some of these species also play pivotal roles in ecosystems, potentially leading to impacts in local to regional ecological function, and compromising overall ecosystem integrity. Animal husbandry operations associated with fiber production also can have environmental impacts through clearing of forests for pasture, which is typically associated with reduced biodiversity, greenhouse gas emissions, soil degradation and reduced water quality and regulation capabilities (Chhabra et al., 2006).

Finally, fibres are vital provisioning services for human well-being, and many livelihoods worldwide are based on the production and trade of fiber goods (Ruiz-Pérez et al., 2004) (section 2.3.5). Fibres are not only important for essential uses, such as clothing and shelter, but also for other non-essential commodities that in many cases are an important component of well-being for many societies, such as elements of the material culture of many traditional and non-traditional groups (Godoy et al., 2005).

2.2.3 Energy

Energy is an important input for the agricultural, industrial and transport sectors and private individuals, constituting an important basis of human well-being. Energy consumption is directly linked to human activities. For example, the amount of energy used by agriculture is increasing worldwide, as mechanization, especially in developing countries, increases. Energy production and consumption vary greatly among and within subregions in the Americas, with the highest level of consumption level occurring in North America (Figure 2.13).

Natural ecosystems provide different kinds of renewable energy, such as heat (e.g. burning of wood or charcoal), electricity (e.g. hydropower) and biomass fuels. Electricity derived directly from natural resources has an extremely high importance in South America, where 81% of produced energy is from renewable sources, mainly hydropower (Table 2.6). In 2011, 55% of the energy matrix of Mesoamerica, the Caribbean and South America came from hydropower (WWAP, 2015). Brazil currently has 158 hydroelectric plants in operation, which total more than 89 gigawatts, with 9 additional plants under construction and another 26 authorized (Tolmasquim, 2016). If micro-hydropower stations are included the number jumps to 1,100 hydroelectric stations (Rocha et al., 2015). Energy production by hydropower is increasing despite substantial controversy over impacts to biodiversity, natural ecosystems, and local populations, including indigenous peoples and local communities (Rocha et al., 2015).

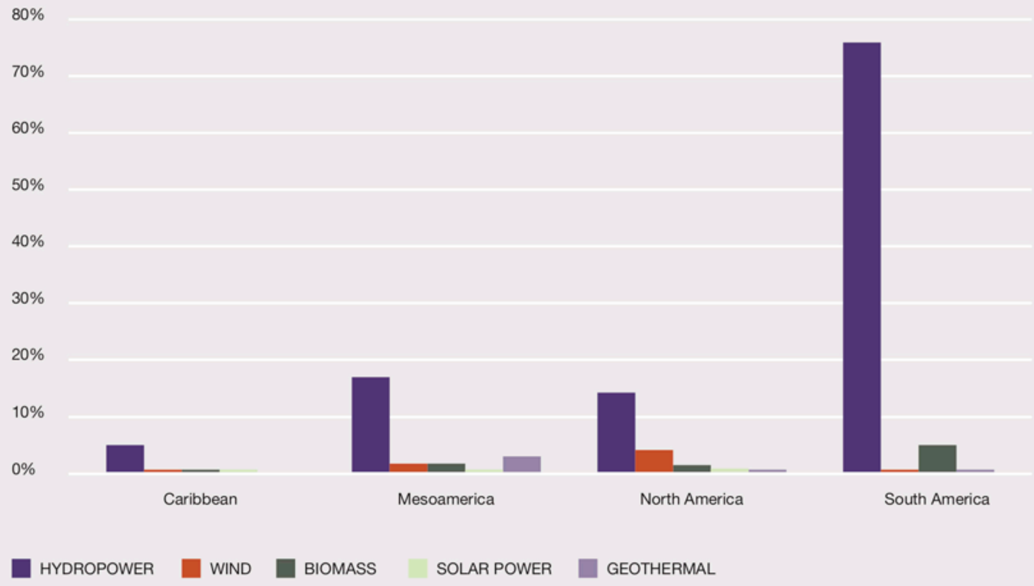
Table 2.6. Per capita annual energy consumption (total kWh/year) in the Americas and percentage (%) of electricity consumption derived from renewable resources by country.

	2003	2013	% Change	% Renewable
North America				
Canada	86,127.7	84,057.2	-2.5	64.5
USA	83,617.1	80,715.6	-3.6	14.3
Mesoamerica				
Costa Rica	11,645.6	12,009.4	3.0	92.2
El Salvador	7,899.9	8,093.6	2.4	60.7
Honduras	7013.7	7,731.6	9.3	45.1
Guatemala	8,318.9	8,958.9	7.1	68.1
Mexico	18,328.0	18,040.4	-1.6	15.7
Nicaragua	6,325.5	6,928.6	8.7	41.1
Panama	12,519.6	12,341.7	-1.4	64.1
Caribbean				
Cuba	11382.7	12,031.3	5.4	4.0
Dominican Republic	9803.9	8,535.5	-14.9	13.8
Haiti	2672.8	4,589.0	41.8	14.5
Jamaica	13190.2	12,646.7	-4.3	9.3
Trinidad & Tobago	18,5725.2	16,9668.6	-9.5	0.2
South America				
Argentina	21,554.7	22,112.2	2.5	31.1
Bolivia	8,605.6	9,167.5	6.1	34.9
Brazil	15,902.4	16,780.8	5.2	84.0
Chile	21,086.6	25,689.6	17.9	37.7
Colombia	8,127.0	7,801.7	-4.2	82.5
Ecuador	9,760.9	11,434.4	14.6	56.2
Paraguay	8,658.2	8,918.5	2.9	100
Peru	7,786.0	8,267.2	5.8	57.6
Uruguay	14,482.3	15,762.1	8.1	63.0
Venezuela	31,145.6	26,507.5	-17.5	66.0

Sources: Total energy consumption from World Bank (2017a) *World Development Indicators*. Indicator: Energy Use Per Capita. https://data.worldbank.org/indicator (Energy & Mining – Energy use). Pecertange of electricity from EIA (2016) during 2012, except USA (data from 2014) and Argentina (data from 2015).

Figure 2 13 Contribution of different renewable energy sources to total electricity production.

Data include: Canada, USA (North America); Belize, Costa Rica, El Salvador, Guatemala, Honduras, Mexico, Nicaragua, Panama (Mesoamerica); Dominica, Dominican Republic, Haiti, Jamaica, Puerto Rico, Saint Vincent and the Grenadines, Trinidad and Tobago (Caribbean); Argentina, Bolivia, Brazil, Chile, Colombia, Ecuador, Paraguay, Peru, Suriname, Uruguay, Venezuela (South America). Year of data is 2012, with exception of USA (2014), Argentina, Brazil and Canada (all 2015). Source: USA Energy Information Administration (2016). International Energy Statistics. https://www.eia.gov/beta/international/data/browser/#/ Data accessed: May 25, 2016.



Biomass fuels are a direct benefit of nature for humans. Biomass can be used for heating (e.g. firewood, charcoal), production of electricity and transportation fuel. There are many techniques to transform biomass into energy (Figure 2.14), and biofuels could be an important energy source in the future, as they have environmental benefits and can provide income for rural populations involved in production (Nigam & Singh, 2011). Even in the highly developed USA, 2.5 million households (2.1%) use wood as the main source for home heating. In an additional 9 million households (7.7%), wood is used as a secondary heating fuel (EIA, 2014: https://www.eia.gov/todayinenergy/detail.php?id=15431). In Brazil, 53% of rural and 5% of urban population rely on biomass as their principal energy source for cooking. The average for the other South American countries is even higher (62% in rural and 9% in urban areas; IEA, 2006). Charcoal remains an important energy source throughout the Americas (Table 2.7), both for household and industrial use (e.g. Brazil, where most of the charcoal is used in industry; GIZ, 2014). While important especially for rural populations with little access to other sources, there may be negative impacts on the environment due to overexploitation, as well as negative effects on human health (section 2.3.4).

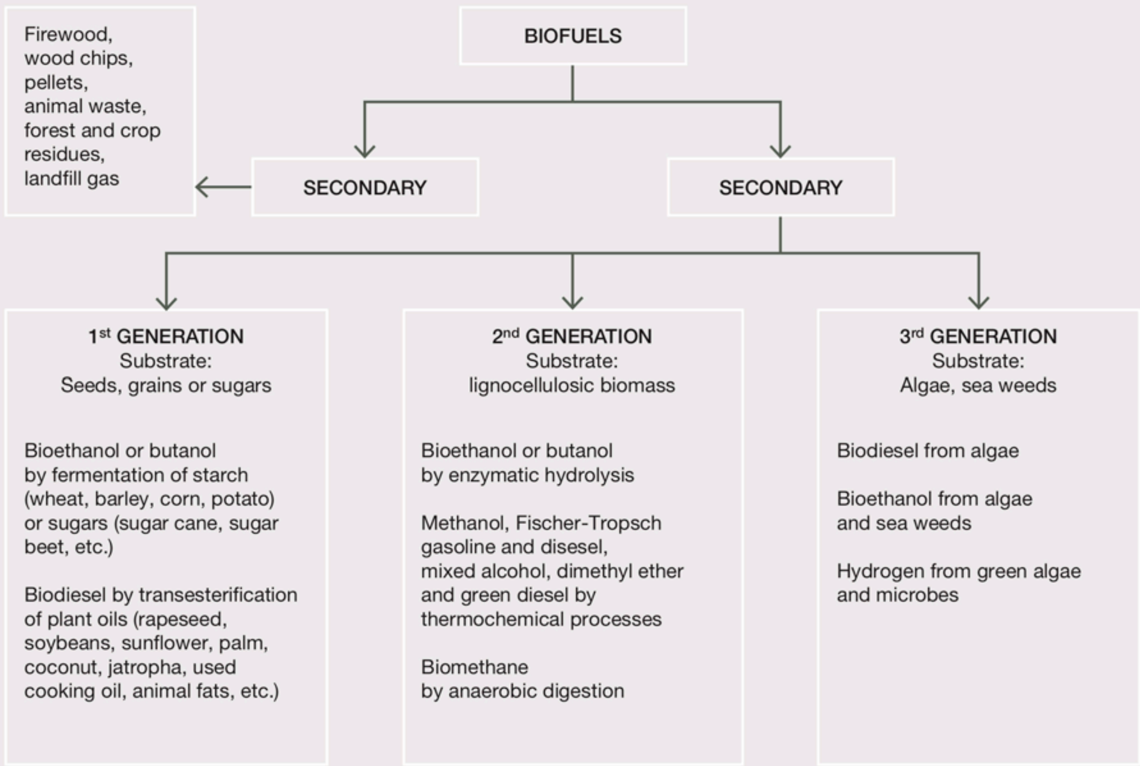
Table 2.7. Charcoal production (tons) in the Americas by subregion

	1986	1996	2006	2016
North America	500,000	789,000	901,800	982,260
Mesoamerica	166,318	117,691	190,742	195,272
Caribbean	115,447	130,704	132,801	177,774
South America	10,779,511	8,,779130	9,532,494	8,283,537
Source: FAOStat (2017). Forestry Production and Trade. http://www.fao.org/faostat/en/#data/FO. Wood charcoal. Last updated December 12, 2017.				

On an industrial level, electricity derived from biomass is increasingly important, especially in South America (Figure 2.13). Brazil particularly uses a great deal of bagasse, produced from sugar cane, that is left over from ethanol production. The use of biomass for fuel production is an important part of the South and North American energy matrix. The USA and Brazil are the world's largest producers of ethanol fuel, 14.8

and 7.1 billion gallons/year, respectively. In fact, the Americas is by far the most important region in the world for ethanol production. Recently, the USA agricultural sector reported significant growth in corn-derived ethanol production, which was encouraged in 2002 by oil price increases and after 2007 by government support policies mandating ethanol use; one negative consequence was a trade-off in which natural habitats were converted to high input agriculture for corn production (Faber & Male, 2012). Gasoline in the USA contains approximately 10% ethanol and in Brazil, 25%. However, due to the large land areas needed for production of first generation secondary biofuels, the biofuels also have the problem of competing for land needed for food supply, necessitating the need for other solutions, for example, by third generation biofuels (Nigam & Singh, 2011).

Figure 2 14 Classification of biofuels. Source: Nigam & Singh (2011).



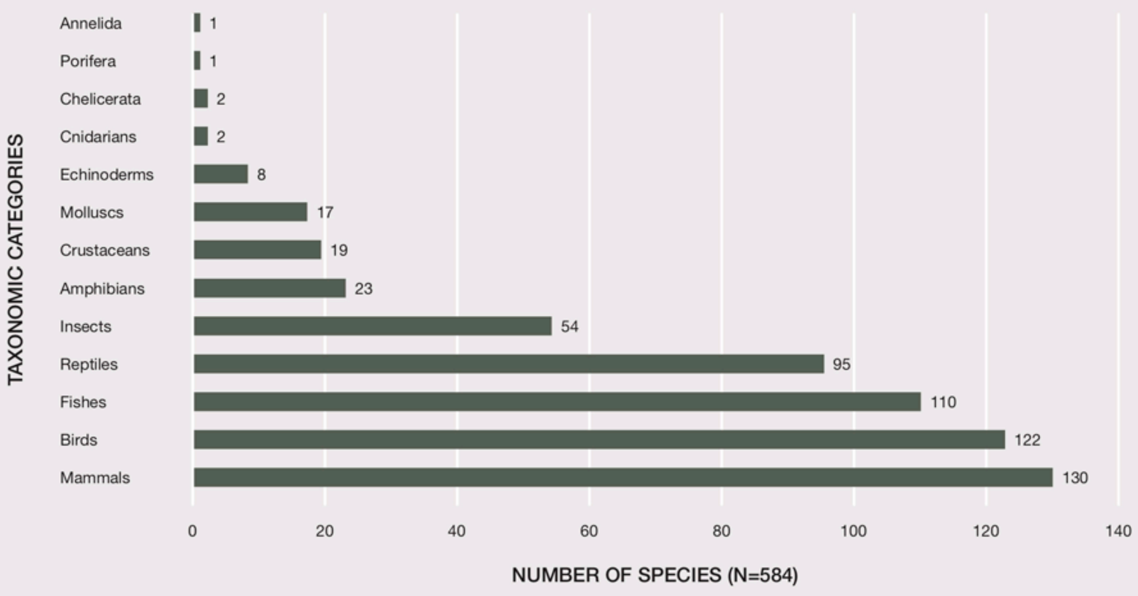
2.2.4 Medicinal, biochemical and genetic resources

Medicines are a crucial NCP derived from biochemical and genetic resources that are obtained from natural and anthropogenic ecosystems, including medicinal plants produced commercially. Between 25-30% of modern medicines come from natural products, including plants, animals and minerals (WHO, 2013). Indigenous peoples and local communities have rich knowledge systems regarding the curative properties of different taxa, as well as the recipes and instructions for their preparation and use. Between 65-80% of the population in developing countries use medicinal plants as remedies (Palhares et al., 2015). Plus, this NCP is intertwined with cultural beliefs and values held by diverse peoples. Plant medicines are made from leaves, roots, flowers, barks, saps and gums, seeds, oils and can be infused in water or oil, ground, used fresh or dried, imbibed, rubbed or inhaled, just to name a few of the diverse ways people use their medicines. The same is true for the medicinal use of animals with products derived from hair, skin, blood, bones, horns, bile, musk, and fats, as well as the whole body of certain insects like ants. Traditional medicines heal physical, psychological and spiritual ills, often without a distinction between them. Although the connection between humans and medicinal plants is long standing, the interest in medicinal products derived from plants has increased since the 20th century. The industrial-scale use of medicinal plants ranges from herbal teas, new drugs, pharmaceutical auxiliary products, health foods, phytopharmaceuticals and intermediates for drug manufacturing (De Silva, 1997). It is estimated that nearly 30% of commercially sold therapeutic medications are derived mainly from plants and microorganisms. In areas such as oncology, this number reaches 60%.

Many local medicinal plants and aromatic herbs used globally are grown in home gardens and not as large scale crops (de Padua et al., 1999). Local and endemic species are almost always connected to a wild harvest while introduced species tend to be used in larger scale productions (Walter & Gillett, 1998). Some herbal supply companies reported to Rainforest Alliance that between 60-90% of their volume of primary material was cultivated. However, this percentage was of only 10 to 40% of the species they use, and the rest were harvested from wild populations (Laird & Pierce, 2002). A total of 546 medicinal plant taxa are used by indigenous peoples of the Canadian boreal forest, from which the most frequently used plant parts are roots, leaves, whole plants, fruits and rhizomes to the treatment of gastro-intestinal disorders, musculoskeletal disorders, cold, cough and sore throat, injuries, respiratory system disorders, urinary system disorders, and dermatological infections (Uprety et al., 2012). In Mesoamerica and the Caribbean region, Alonso-Castro et al. (2016) documented 104 plant species belonging to 55 families that are used as immune-stimulants, of which only 27% have been the subject of pharmacological studies. Kujawska et al. (2017) registered 509 botanical species used as medicinal plants in Argentina, comparing their use by three cultural groups of people: Guarani Indians, Criollos (mestizos), and Polish immigrants. The Guarani were the most expert in medicinal plants, using the greatest diversity of species (n=397). Polish immigrants used the least (n=137), in part due to the challenges of establishing a new pharmacopedia in their new, highly diverse environment. In the tropical Atlantic forest of Brazil, Di Stasi et al. (2002) documented a pharmacological inventory with people from rural and urban communities that includes 290 herbal remedies prepared from 114 medicinal plants cited for 628 medicinal uses. Clearly the Americas host a large percentage of the world's 28,187 known medicinal plants (Willis, 2017). For example, at the country level, numerous reports show the high levels of medicinal plant biodiversity used in the Americas, including 2,500 in the USA (Moerman, 1996), 4,000 in Mexico (Caballero et al., 1998), 5,000 in Colombia (Fonnegra & Jimenez, 2007), and 1,529 in Argentina (Barboza et al., 2009).

In addition to medicinal plants, animal-based remedies (zootherapy) are found in the all the Americas subregions, mainly used by indigenous peoples and local communities. Alves & Alves (2011) reviewed the literature from Latin America and found that at least 584 animal species, distributed in 13 taxonomic categories, are used in traditional medicine (Figure 2.15). The use of wildlife as medicine represents not only an economic benefit from sales or by saving money for families, but it constitutes a knowledge and value system tied to inheritance, belonging, and identity.

Figure 2 15 **Number of animal species used as medicinal remedies in Latin America, organized by taxonomic groups. Source: Alves & Alves (2011).**



International trade in medicinal plants is expanding (Table 2.8), and exports are largely in an unprocessed or slightly processed form with much of the economic return going to intermediaries. As a consequence of this increase in economic activity, the monetary value of medicinal plants has also grown. In 2000, $17 billion was spent in the USA on traditional herbal medicines. In 2002, the annual global market for herbal medicines was estimate to be worth $60 billion (WHO, 2002) and by 2012 the global industry in traditional chinese medicine alone was reported to be worth $83 billion (Royal Botanical Gardens Kew, 2017) Still native chemodiveristy is an almost untapped source of economic development with a very low environmental impact, since once isolated and tested the new compounds are synthesized to be produced in the scale needed for a new medicine. New compounds can also be important for the food and for the agrochemical industry (Kalin-Arroyo et al., 2009; Desmarchelier, 2010; Joly & Bolzani, 2017). Furthermore, the advent of genetic techniques that permitted the isolation/expression of biosynthetic cassettes from microbes may well be the new frontier for natural products lead discovery. It is now apparent that biodiversity may be much greater in those organism, and the numbers of potential species involved in the microbial world are many orders of magnitude greater than those of plants and multi-celled animals.

Table 2.8. Volume of medicinal plant exports from the Americas by subregion and country.

Region/country	Exports volume of medicinal plants					
	1991	1994	1996	1998	2000	2002
Global	**371.9**	**449.4**	**489.0**	**463.7**	**529.1**	**583.6**
North America	**8.0**	**13.8**	**15.3**	**19.3**	**19.7**	**15.7**
USA	7.7	13.2	14.0	17.4	18.0	12.6
Canada	0.3	0.6	1.3	1.9	1.7	3.1
Mesoamerica and the Caribbean	**8.1**	**5.9**	**15.9**	**14.9**	**43.1**	**131.0**
Mexico	8.0	5.2	15.1	13.9	42.6	130.2
Dominican Republic	0.0	0.0	0.0	0.0	0.1	0.2
Costa Rica	0.0	0.1	0.2	0.1	0.1	0.1
South America	**16.4**	**16.5**	**20.1**	**23.2**	**17.4**	**20.9**
Chile	9.7	10.4	13.7	15.8	9.9	10.0
Argentina	3.2	2.3	2.4	2.2	1.6	1.2
Peru	1.5	1.8	1.6	1.7	2.4	3.4
Brazil	1.0	0.9	1.2	1.7	1.8	2.0
Bolivia	0.6	0.5	0.6	0.6	0.3	0.4
Ecuador	0.1	0.2	0.2	0.2	0.8	2.3
Source: FAO (2002)						

Indeed, the history of peoples in the Americas is intimately linked to the land, water, plants and wildlife, including medicinal uses of these (Toledo & Barrera-Bassols, 2008). However, medicinal species are being harvested at ever-expanding volumes to fulfill the regional and international demand, mostly from the wild (Kuipers, 1997; Lange, 1998). The technical advances in the pharmaceutical industry permit synthesis of some active compounds, but of the 45 plant-based drugs developed from tropical rain forest species in the 1990's none is known to be synthesized (Farnsworth & Soejarto, 1991). Efforts at synthesis of the phytochemical complexity of tropical plants have not been economically successful, thus companies require natural sources of raw materials (Laird, 1999). Therefore, the degradation and transformation of natural habitats affects this NCP negatively impacts the primary health care option of millions of urban and rural citizens (Shanley & Luz, 2003). In this context, the consequence of biodiversity loss affects not only potential research into the pharmaceutical benefits of these species, but is particularly devastating to those people without access to western medicines (Box 2.3).

Box 2.3 Traditional medicine in Cerrado, Brazil. Source: Dias & Laureano (2017).

In the Brazilian Cerrado (savanna-type biome), *raizeiras* (local healers –mainly women - and midwifes) use a diversity of medicinal plants to treat the ailments of rural people. In the state of Minas Gerais alone, 264 different medicinal plants are used by *raizeiras*, 40% of them being wild plants (Dias & Laureano, 2010). *Raizeiras* organize themselves in "community pharmacies" to produce medicine to be sold locally, where there is no access to state-supported medical nor conventional drugstores. *Raizeiras* are able to identify the causes of illness, whether congenital, socioenomic, endemic or mental illness (often related to spiritual causes). Local medicines are usually imbued by values attributed to faith and spirituality, including prayers, religious rituals, and indigenous local knowledge. Also, such knowledge to collect and manage plants in ways that conserve them for future generations. Nevertheless, the conversion of Cerrado vegetation due to agribusiness expansion and the restriction of access to previously commonly-held land are current threats to these "community pharmacies," putting at risk the health of thousands of people in central Brazil. This is one of many of examples from Mesoamericaand South America; see: http://www.biodiversidad.gob.mx/Biodiversitas/Articulos/biodiv62art3.pdf

2.2.5 Learning and inspiration

Landscapes and seascapes, whether natural or transformed by anthropogenic activities, as well as biotic organisms, provide opportunities for learning and inspiration for humans in all biomes and subregions of the Americas. Indigenous language, knowledge and practices, as well as local farm knowledge and practice are transmitted through living in nature. Fishing and hunting knowledge too are transmitted through practice. In the USA alone, each year an estimated 29.6 million people over 16 years of age fished in freshwater ecosystems for a total of 463 million days, another 8.9 million people over 16 years of age fished in marine environments for a total of 99 million days, and over 2.5 million people over 16 years of age hunted for migratory birds (mostly waterfowl) for a total of 23 million days. Plus, about 22.5 million (9% of all people over 16 years of age) travel away from home to watch wildlife, and 45 million actively observed biodiversity around the home (e.g. bird feeders) (FWS, 2011).

In a global review of the strong benefits that interacting with nature has on cognitive ability and function, Keniger et al. (2013) identified the following benefits: attentional restoration, reduced mental fatigue, improved academic performance, education/learning opportunities, improved ability to perform tasks, improved cognitive function in children, and improved productivity. In urban settings, the restorative benefits of a view of nature, even if only from a window, has been documented (Kaplan et al., 2001). Keniger et al. (2013) argue that there is good evidence that exposure to nature in both urban and wilderness settings may improve cognitive performance, as demonstrated by studies in Michigan (Berman et al., 2008) and in California (Hartig et al., 1991). On the other hand, Russell et al. (2013), in a review on how knowing and experiencing nature influence human well-being, have discussed the relative lack of empirical studies regarding the effects of nature on learning.

Some religions make use of plants and animals to connect humans to the spiritual world. For instance, the use of *ayahuasca*, a drink made of two Amazonian plant species (*Banisteriopsis caapi* and *Psychotria viridis*) has become more and more popular through the *Santo Daime* religion in many urban centers in parts of South America (Labate, 2004). There are very few studies that investigate the role of nature as spiritual inspiration for non-indigenous people in the Americas. Fredrickson and Anderson (1999) claim that outdoor recreational trips act as spiritual inspiration for women experiencing wilderness in the USA, and in many coastal and rural communities of the Americas, people with no access to weather forecast “read nature’s signs” to plan their planting, harvesting or fishing activities. Nature is also an unlimited source for scientific research, and environmental education programs are growing, often with the goal of increasing ecological literacy (McBeth, 2011). In urban areas, green space, zoos, aquariums and botanical gardens are all facilities to promote learning experiences for people. In the USA, more than 183 million people visit aquariums and zoos annually (AZA, 2017). Additionally, a large portion of artwork produced by humankind is inspired in nature, in particular those produced by indigenous peoples and local communities.

Quantifying how much learning and inspiration from nature contribute to human quality of life is not a trivial task. However, one way to assess how the Americas’ peoples value nature for its power to inspire is through institutions that establish rights to relational ecosystem values, as the case from the USA illustrated in Box 2.4.

Box 2.4. Institutions to establish rights to relational ecosystem values.

Just as institutions and governance systems exist to manage instrumental values of ecosystems – the benefits people receive from nature – so too do they exist to manage relational values. The USA provides two prominent examples of national laws established to protect the relational values required for human well-being, living in balance with nature, and spiritual fulfillment. The first is the National Park Service Act of 1916, which established a federal agency to manage areas of extraordinary natural and historical importance to people. Franklin Lane, Secretary of the Interior at the time of the establishment of the National Parks Service, described its lands as “set apart for the use, observation, health, and pleasure of the people.” Nearly 50 years later, the Wilderness Act was passed in 1964 to preserve and ensure continued, but limited access by people to areas “in such manner as will leave them unimpaired for future use and enjoyment as wilderness....” In signing the Wilderness Act, President Lyndon Johnson expressed the purpose of the law as maintaining human well-being through a relationship with nature, indicating that "... once man can no longer walk with beauty or wonder at nature, his spirit will wither and his sustenance be wasted." Together the National Parks Service and the Wilderness Preservation System protect nearly 80 million hectares of wild lands and sites of historic or spiritual significance and can serve as a model for institutional approaches to ensuring the provision of relational services. The clear purpose of these laws is to preserve current and future access to relational ecosystem values, including protecting places of special significance to people and providing assurance that millions of hectares are available to maintain peoples' basic connection with nature for learning, inspiration and other non-material NCP.

Another way to assess how people value nature is through the impact of losing it. In the Americas, most cities are growing at the expense of agricultural areas (HABITAT, 2012), leading to cultural transformations, such as the loss of knowledge and appreciation for native biodiversity that is also linked to rituals and other cultural uses. In fact, globally the capacity of ecosystems to provide cultural services have strongly decreased in the past century (MEA, 2005). For example, the direct degradation of the environment or the decoupling of the ways of living in a habitat also harm sense of place, language diversity and local ecological knowledge (Rozzi et al., 2006, 2012). This is linked to the fact that 61% of the native languages of Americas are either in trouble or dying (Table 2.1 , Simons & Fennig, 2017).

2.2.6 Supporting identities

Nature supports human identities by providing materials and physical places that in turn are part of symbolic and social relationships that form cultural identities. For example, in the Bolivian Andes, the maintenance of well-organized ancestral indigenous agriculture and llama herding emphasizes the respectful use of the environment, conceived of as Mother Earth (or Pachamama) (Choque, 2017). Indeed, nature provides the basis for religious and spiritual experiences in many cultures. In Brazil, Fernandes-Pinto (2017) registered over 400 sacred natural sites representing a variety of ecosystems (e.g. streams, forest, coastal habitats) associated with a diversity of cultures and religions. The author also observed the religious use of public lands in over 100 Brazilian protected areas. Ecuador has recognized the important link of biodiversity and local culture by declaring "Intangible Areas" (like the Tagaeri Taromenane of Yasuní Biosphere Reserve), which are large extensions of biodiverse territory where indigenous peoples want to be isolated from western culture. This is one of the best examples of zoning protected areas that take into consideration the relationships between nature and society, with legally functioning frameworks (http://wrm.org.uy/es/articulos-del-boletin-wrm/seccion1/ecuador-la-zona-intangible-tagaeri-taromenane-del-yasuni/).

Material NCP, like food, also contribute to the cultural identity of indigenous people and local communities. For example, apart from food, North American indigenous peoples value wildlife as an integral part of their way of life and many follow complex rituals, which guide their relationship with their subsistence species, including identity in clan names, oral histories (Erdoes & Ortiz, 1985), ceremonial preparation for hunting and cooking, transformation and spiritual communication (http://www.traditionalanimalfoods.org) (Kuhnlein & Humphries, 2017). In biomes like the Canadian tundra (Kuhnlein & Chan, 2000; Usher, 2002), local economies are made up of a mix of cash and subsistence, depending strongly not only on the availability of local resources, but also on cultural knowledge, traditionally transmitted from generation to generation, regarding the ways of preparation, storage, and distribution of food and resources. Therefore, Inuit identity is supported by their environment and the traditional cultural practices conducted in it, especially hunting, and in this sense, the consumption of wild animal meat is vital not only for Inuit health, but also their identity. Within the Inuit knowledge and value system, hunted animals, such as seals or polar bears, and humans are linked together in a spiritual relationship that both depend upon (Borré, 1991; Dowsley, 2010; Fialkowski, 2012). Among the Quileute this physical-spiritual connection is acknowledged by throwing the bones and head of the first salmon caught back into the river to ensure good will of the salmon spirits. This was also meant to symbolize taking only what was needed, but served as a reminder to strive for balance (Fialkowski, 2012).

While attempts at monetization of ecosystem services may lead to some insights on the values of nature, broader considerations related to spirituality, cultural identity or social cohesion are not easily characterized in this value system, making them too often underrepresented in decision making and in scientific assessments at subregional and regional levels. Recent approaches to integrate social and ecological factors, which can help to identify the instrumental, intrinsic and relational values of nature, could improve attention to cultural and identity in the long-term (Chan et al., 2012). Notwithstanding the lack of systematic data on status and trends, it is well established that nature substantively supports such economic activities as hunting and fishing. In turn, hunting is inextricably related to leadership building, territorial control, and cultural stories (Townsend & Macuritofe-Ramírez, 1995; Erdoes & Ortiz, 1985; Urbani, 2005; Urbani & Cormier, 2015; Cormier & Urbani, 2008), art (Salinas, 2010) and rituals (Baleé, 1985) of indigenous peoples and local communities throughout the Americas. Fishing too is valued for its contributions to food and livelihood securities, and like hunting and fishing practices also connote cultural values that have to do with a "way of life," cultural continuity, knowledge systems and connections to place (e.g. Trimble & Johnson, 2013).

There is strong evidence that both species and cultural diversity are decreasing in the Americas (see Chapter 3 and section 2.1) and changes in development models that act as drivers (see Chapter 4) also lead to an erosion of nature's support for identities and this trend is increasing. Drivers of such change include internal migration (e.g. rural to urban), cultural assimilation, restricted access to nature (section 2.5), limiting the practices and relationships with nature, which are the constituents of cultures and identities. For instance, tropical dry forests are valued in additional ways aside from a utilitarian approach based on economic market values of goods and services provided (Birch et al., 2010; Castillo et al., 2005; Maass et al., 2005). Socio-cultural values in these forests are particularly important for many traditional and indigenous populations whose identities, worldviews, cosmologies and traditions are closely linked with particular characteristics and conditions of these ecosystems (Balvanera et al., 2011). In turn, identity and culture of a place can feed back into well-being via other mechanisms, like tourism, as many people visit such places for aesthetic enjoyment and spiritual fulfillment. These services are also less emanble to pecuniary valuation methods than provisioning services or material NCP, yet in many instances represent a key factor for good social relations.

Erosion of nature's support for identity has a direct effect on well-being. For instance, in Canada, loss of cultural identity has impacted the mental health of the First Nations, Inuit, and Métis, leading to high rates of depression, alcoholism, suicide, and violence in many communities, with the greatest impact on youth (Kirmayer et al., 2000). Many First Nations youth are unable to take on their traditional cultures because so many practices have been restricted by losing access to traditional lands. For example, many tribes in the USA plains states that revere the buffalo (*Bison bison*) for its power and the good fortune the buffalo spirit brought to the tribe, no longer have access to the animal. The eroded cultural identity associated with losing access to traditional lands has meant that many indigenous people now suffer from chronic socio-economic problems (Carpenter & Halbritter, 2001). Unfortunately this trend is also observed among other American indigenous peoples. For instance, the suicide rate among Guaraní Kaiowá and Nandeva youth in Brazil is higher than the national average, and the rate appears to be increasing among young males (Coloma et al., 2006).

2.2.7 Physical and psychological experiences

Literature reviews on how knowing and experiencing nature influences human well-being have clearly shown its benefits on mental and physical health. Russell et al. (2013) conclude that "the balance of evidence indicates conclusively that knowing and experiencing nature makes us generally happier, healthier people." Conversely, experiencing the loss of an ecosystem service, led respondents to report that their emotional, psychological, or spiritual well-being is harmed; highlighting the importance that nature has on their quality of life (Federal Provincial and Territorial Governments of Canada, 2012). Relative to other non-material NCP, there is a large amount of literature linking nature with well-being through increased health benefits. This is particularly important in urban environments, where increasingly larger proportions of people live (Table 2.9), and where stressors like increased noise, over stimulation, and health problems derived from sedentary lifestyles are frequent. For example, a recent and exhaustive review on the benefits of interacting with nature presents a wide range of studies demonstrating benefits to physical health, cognitive performance and psychological well-being with fewer, studies reporting on social cohesion and spiritual benefits (Keniger et al., 2013). These same authors showed that studies on the benefits people receive from interacting with nature have a regional bias towards Western developed nations with 79% of the 59 studies assessed reporting results from North America and Europe and none for South America and Africa. The authors conclude that, although a broad range of benefits that accrue from interacting with nature have been described, most of the evidence is descriptive. Therefore, less is known about the mechanisms by which benefits are delivered, the characteristics of natural settings and how these characteristics may affect the resulting benefits in different geographical locations, cultures and socio-economic groups. This complexity is important to understand, though, to improve urban and regional planning that enhances well-being through nature interaction. This is particularly vital as most subregions in the Americas are strongly urbanized and this trend is increasing (see also Chapter 1) so that the large majority of the population has limited access to natural or wild landscapes and seascapes. Although information is limited, the valuation of natural areas has been more studied in Northern Hemisphere cities (see Niemela et al., 2010), where nature changes dramatically year round in contrast to countries located in tropical regions. Low valuation of nature and/or the loss of human-nature interactions can have serious consequences on people, not only in the decrease of benefits derived, but also the disconnection from nature that results (Soga & Gaston, 2016). This can decrease favorable attitudes and behavior, decreasing motivation to protect it (Lopez-Mosquera & Sanchez, 2012; Dallimer et al., 2014).

Table 2.9. Proportion and annual rate change in urban population in the Americas by subregion.

Subregions	**% of urban population**			**% annual rate change in urban population**
	1990	**2014**	**2050**	**(2010-2015)**
North America	75	81	87	0.2
Mesoamerica	65	73	82	0.4
Caribbean	58	70	81	0.8
South America	74	83	89	0.3
Source: United Nations Population Division (2014).				

One way to assess physical and psychological experiences with nature is through nature-related tourism assessments (Table 2.10). Nature-based tourism generates both livelihoods and income for providers, ranging from small rural communities and protected areas to large coastal resorts. Overall, tourism is a major resource for many economies in mountainous areas, and studies also have shown that protection of watersheds provides greater economic value than resource extraction (The Mountain Institute, 1998; UNEP, 2008). In addition, it generates leisure experience for costumers. For instance, in addition to being associated with export earnings, coffee plantations provide cultural services and earnings from agrotourism activities in places like Mexico and Guatemala (Lyon, 2013). Protected areas also provide income through jobs and park fees. For example, some important national parks in the USA are located in the Rocky Mountains (e.g. Yellowstone, Grand Teton, Glacier, and Rocky Mountain National Parks) and protect outstanding examples of mid-latitude alpine and subalpine environments in North America (Funk et al., 2014). Among Brazil's national parks, 20 receive more than 10,000 visitors per year (data for 2013; de Castro et al., 2015). Numbers of visitors are primarily determined by the natural beauty of the region and by the variety of opportunities for recreation and associated services and infrastructure (de Castro et al., 2015). In 2013, Tijuca National Park, situated within the city of Rio de Janeiro, Brazil, and Iguaçu (Brazil) and Iguazu (Argentina) National Parks, with their famous waterfall, received between 1.5 and 2.8 million visitors, respectively (data only for the Brazilian portion of Iguaçu). Together this accounts for 74% of total visitation in Brazilian national parks. Interestingly, the forests in Tijuca National Park are actually the result of a reforestation program in the 19th century, when more than 70,000 trees were planted to protect local water resources with their high importance for Rio de Janeiro.

Table 2.10. Examples of economic valuation of the nature-related tourism sector.

Winter tourism (skiing) industry in USA for 2009-2010	$12.2 billion	Burakowski & Magnusson (2012)
Tourism in the Caribbean	$28.4 billion (13% of GDP)	CARSEA (2007)
Coral reef associated tourism and fishery in St. Maarten	$57.6 million	Bervoets (2010)
Coral reef associated tourism/recreation in Tobago for 2006	$101–130 million	*Kushner et al*. (2011)
Coral reef associated tourism/recreation in St. Lucia for 2006	$160–194 million	*Kushner et al*. (2011)
Sport fishing and waterfowl hunting on 1.3 million acres in coastal Louisiana, USA	$272 million (converted from 1990 US $)	Bergstrom *et al*. (1990)
Maya Biosphere Reserve in the Petén area of Guatemala	$47 million and provides employment to 7000 people	CBD (2008)
National protected areas in Costa Rica	$1.3 billion in 2009 (~5% of GDP)	Moreno (2011)

The Caribbean's islands are more dependent on income from tourism than that of any other part of the world, accounting for 15.5% of total employment (CARSEA, 2007). In 2015, about 9 million tourists visited the subregion (CTO, 2015). In 2013, international tourism receipts were 45% of total exports. For example, the earnings from tourism were more than 80% of total service exports in The Bahamas and Saint Lucia, and more than 70% in Aruba, the Dominican Republic, Grenada and Jamaica (IDB, 2016). The tourism sector has required large investments in coastal development to cater for the high influx of tourists and the associated demand for hotels, marinas, harbors, shops, and sports facilities. These rapid developments have had major impacts on the coral reefs; with 32% of Caribbean coral reefs estimated to be threatened by coastal development (Bryant et al., 1998). Additionally, in many areas the sheer numbers of dive and snorkel tourists cause direct damage to coral reefs. Average coral cover declined by more than 50% from 1970 to 2011, but the disparity among locations was great (Jackson et al., 2014).

Nature contributes to tourists indirectly through the benefits gained from the recreational experiences. Based on USA expenditures for sport fishing and dock-side expenditures for commercial fish, the benefits from sportfishing rival the food and raw material benefits from commercial fishing. In the USA alone in 2010, an estimated 33 million people over 16 fished for sport for a total of 554 million days supporting over 820,000 jobs and $35 billion in salaries and wages (FWS, 2011). Over 80% of USA anglers fished in freshwaters (FWS, 2011), spending in total, > $47 billion on the sport. The $4.9 billion spent by salt-water anglers in 2014 rivaled the $5.5 billion for dockside purchases of commercial fish (NMFS, 2016), which is an important trade-off consideration in fishery management decision-making. Since the tourist industry benefits from greater recreational expenditure and the recreationists benefit from a less expensive satisfying experience, there is an optimum balance of costs and benefits that maximizes benefits across both groups.

2.2.8 Maintenance of options

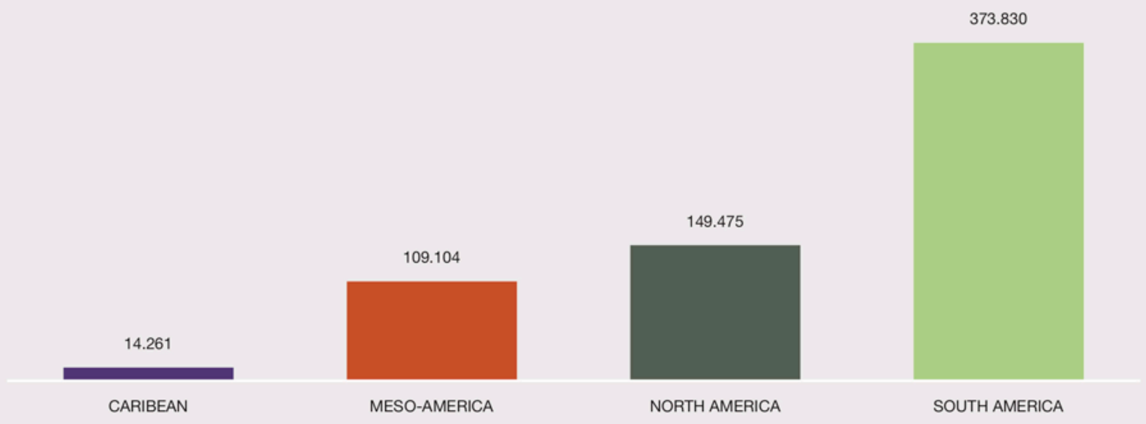
An overarching benefit of ecosystems is their ability to provide services and maintain options for a good quality of life for both present and future generations. These options are sustained by biodiversity and are lost as biodiversity is eroded. Since the future is uncertain, any loss of the irreplaceable attributes of nature diminishes the potential for improved quality of human life. Future options can be maintained by either protecting species from loss or by setting aside areas that support the diversity of ecosystem elements in all their characteristic complexity. Species protection through various laws like the USA Endangered Species Act and the multi-lateral Convention on International Trade of Endangered Species (CITES) work toward this end. But the strategic protection of land and water from destructive use may be the most widely advocated policy instrument to maintain options sustained by biodiversity, including ecosystem restoration where needed. These and other instruments are described in Chapter 6, including ecosystem restoration.

Most nations in the Americas now recognize the value of protecting critical geographical areas and threatened species from consumptive use and to maintain ecosystem functionality through sustainable use of ecological resources. Furthermore, establishing protected areas for restoration of key resources for local communities has also been demonstrated to provide important benefits (e.g. Aburto-Oropeza et al., 2011). The proportion of land and marine areas with valuable ecological resources now claimed to be protected from destructive use is 14.8% worldwide ranging from 11.6 % in North America to 23.3% in Latin America and the Caribbean (World Bank Database, 2017b, http://data.worldbank.org/indicator/ER.LND.PTLD.ZS). There is also recognition that indigenous lands may be a powerful instrument for protecting nature, and initial estimates suggest that at least 1.2% of the land area of North America, 19.5% of Mesoamerica, and 10.5% of South America are protected through this designation (see Chapter 1, section 1.6.2).

Not all biomes are equally protected, however. Portillo-Quintero and Sánchez-Azofeifa (2010) found only 0.3% of the total area with tropical dry forest is under some category of protection in Mesoamerica, ranging from less than 0.4% in Mexico and El Salvador to 15% in Costa Rica. About 12% of the northern temperate forests of Canada are under protection and about 6% of all forest is being sustainably managed as described by the Forest Stewardship Council (https://ic.fsc.org/en). In the southern temperate forest ecoregion, extensive protected areas are owned by private individuals, NGOs and governments (Soutullo and Gudynas 2006, Rozzi et al. 2012). For example, 57.3% of the Magallanes Region in southernmost Chile is under government protection (SIB Magallanes, 2017). Grasslands and savannas cover 30% of the Americas' terrestrial surface, span from South to North America, and cover a broad altitudinal gradient. Nevertheless, they are a poorly protected biome, primarily because grasslands have experienced extensive transformation for agriculture production. Of the five biomes in Brazil (rainforest, dry woodlands, savanna, grassland and wetlands), the Pampas grassland biome has the highest Conservation Risk Index (Overbeck et al., 2015). Ecological restoration and rehabilitation are likely to be essential strategies for maintaining endangered options in these highly degraded ecosystems (Galatowitsch, 2012).

Protected areas help to avoid habitat degradation and loss of biodiversity and so make significant contributions to providing a variety of NCP (Bruner et al., 2001; Dudley et al., 2007; Andam et al., 2008; Leverington et al., 2010; Joppa et al., 2008; Nagendra, 2008; Nelson & Chomitz, 2011; Ferraro & Hanauer, 2011). The capacity to maintain NCP is highly correlated with management ability and investments (Dudley, 2008) (see Chapter 6). In the USA, for example, ~28% of federal public land area is managed for multiple uses, including protection of threatened species and their habitats (Bowes & Krutilla, 1989). World protected areas are estimated to store over 312 gigatons of carbon or 15% of the terrestrial carbon stock (Kapos et al., 2008). Marine protected areas have proven to be effective at preserving biodiversity, but differing views on their goals have resulted in conflicts about whether to manage for preservation or integrated sustainable use (Agardy et al., 2011). Wetlands (both coastal and inland) identified as sites of international importance by the Ramsar Convention are the focus of national and international cooperation for the conservation of biodiversity and are managed for sustainable or "wise-use" by fostering wetland dependent human activities and livelihoods, for example food production (such as wild rice, waterfowl), the regulation of water supplies, tourism and education. As of 2016, there were nearly 650,000 km^2 of wetlands identified as internationally important in the Americas (Figure 2.16; http://www.ramsar.org/about/wetlands-of-international-importance-ramsar-sites).

Figure 2 16 The area (hectares) of wetlands in the Americas designated under the Ramsar Convention as wetlands of international important, by subregion. Source: www.ramsar.org. Data accessed: March 17, 2017.



In the Americas, the areas protected by law or other official action increased rapidly between 1970 to 2010 to about 17% of the total area identified as key biodiversity areas by conservation organizations, but has slowed down more recently (see Chapter 3). The creation of new protected areas peaked between 1980 and 2000 in North America and the Caribbean, and since 2000 in Mesoamerica and South America (Figure 2.17). In North America, the fraction of land area protected (11.6%) is less than the fraction of territorial marine waters protected (16.4%). In Mesoamerica, the Carribbean and South America, the fraction of land area protected (23.5%) exceeds the fraction of territorial marine waters protected (15.5%) (World Bank, 2017c, https://data.worldbank.org/indicator/ER.PTD.TOTL.ZS?view=chart).

Figure 2 17 **Protected areas in the Americas region, 1850–2016.**

Source: Protected Planet (2017). Protected area coverage per country/territory by UN Environment Regions. https://protectedplanet.net/c/unep-regions. Date accessed: March 19, 2017 and Brooks *et al.* (2016a and 2016b).

Present and future land and water protection and restoration could moderate some potential effects of future climate change by protecting and enhancing vital NCP, such as the regulation of water flow and quality, feeding and nursery areas for fisheries, wildlife and other species on which human societies depend, resistance to invasive alien species, coastal erosion protection, reservoirs of wild crop relatives, and carbon storage (World Bank et al., 2010). In the Brazilian Amazon, for example, protected areas and reserves for indigenous peoples could prevent an estimated 670,000 km² of deforestation by 2050, representing 8 billion tons of avoided carbon emissions, contingent on effective management across diverse jurisdictions including state, private sector, indigenous groups and local communities (Dudley et al., 2009). But adaptation to climate change could also require even greater protection and restoration to conserve and recover required connectivity in areas where biomes and ecosystems are highly fragmented. Despite the conservation benefits, the establishment of protected areas requires consideration of trade-offs associated with potential negative impacts on local livelihoods and well-being (e.g. displacement, restricted access to medicinal plants and animals as well as food and sacred sites). Pullin et al. (2013) performed a systematic review of the impacts of protected areas on human well-being globally, using cases from North, Mesoamerica and South America and found that the existing evidence is inconclusive about the best way to inform policy makers about win-win solutions for promoting both NCP and quality of life.

2.2.9 Climate regulation

Many ecosystems are effective at taking up and storing carbon, thereby helping to regulate climate. Carbon uptake and storage helps mitigate the accumulation of greenhouse gasses in the atmosphere that result from fossil fuel combustion (270±30 Pg C released since the Industrial Revolution), land use change (136±55 Pg released from deforestation, biomass burning, wetland drainage and conversion to agriculture), and soils due to land degradation (78±12 Pg C) (Lal 2004). Micro-climate regulation is facilitated by the presence of natural vegetation that helps modify temperatures and soil water content, due to the effects of vegetation on albedo (the ability of an area to reflect solar energy). It should be noted that while the use of fossil fuels is changing the biosphere it also has contributed to improvements in human health and prosperity (Costello 2009).

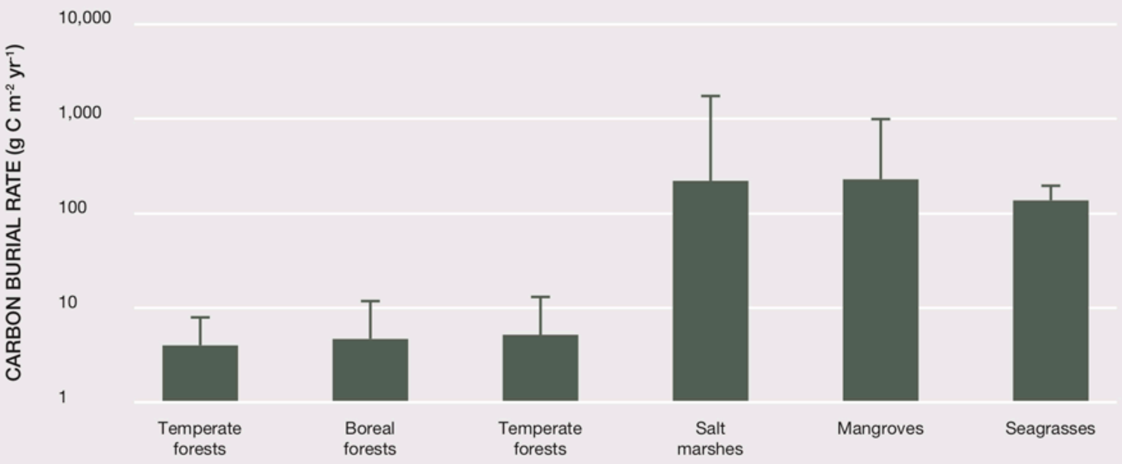
Human well-being is directly and indirectly linked to climate. For instance, the redistribution of species (i.e. latitudinal shifts) predicted to occur in response to climate change is expected to have far reaching effects with changes in agricultural production and the abundance of species that many people rely on for food (e.g. species of fish, crops, pollinators), impacting food security (Pecl et al., 2017). Shifting climate zones will affect local communities' use of traditional travel routes where, for example, they pursue activities such as reindeer herding, hunting, and berry harvesting. The range of disease carrying organisms will also change, potentially introducing vector-borne diseases to new areas with increased incidence of virus transmission like malaria and Zika (Pecl et al., 2017). Generally, climate change poses a threat to water and food security, and is expected to lead to an increase in extreme events that may, in turn, cause human migration ('climate refugees') from storms, floods, and wildfires, particularly in urban settings (Patz et al., 2005). The impacts of climate refugees for countries in the Americas will be felt both for those sending out and receiving refugees. Increases in human mortality rates between the mid-1970s and 2000 due to climate change are estimated to range from 0-70 deaths per million people in the Americas; this risk is projected to more than double by 2030 (Patz et al., 2005). Ultimately, the impact of climate change is expected to be largest for populations with limited access to resources and who have contributed little to its cause (Costello et al., 2009), thereby invoking issues related to environmental justice. For its part, biodiversity can modify both exposure and impacts from extreme events.

In general, carbon dioxide uptake through plant photosynthesis is the world's greatest carbon sink. Forest growth can store carbon dioxide for up to 800 years; rates of carbon uptake vary with climate and can also increase due to the deposition of atmospheric nitrogen, which acts as a fertilizer and increases tree growth rates (Luyssaet et al., 2008). The Amazon rainforest alone is estimated to hold 90,140 billion metric tons of carbon (Fauset et al., 2015). Deforestation is a leading contributor to climate change, responsible for an estimated 15% of global greenhouse gas emissions. Millions of hectares of tropical forests are cleared annually, primarily for agriculture (small scale and large scale farming and grazing) (FAO, 2016), and the loss of soil carbon in cleared areas can amount to 40% over the first 5 years (Detwiler, 1986). In cases where harvested wood is used to make consumer products, carbon is stored through the life cycle of the wood, and subsequent forest regrowth sequesters additional carbon, mitigating greenhouse gas emissions (Smyth et al., 2014). Recovery or restoration of degraded and deforested lands can increase carbon uptake from the atmosphere, increase the flow of ecosystem services, and help alleviate rural poverty (Lamb et al., 2005).

Globally, soils are a major reservoir of carbon, second only to that which is held in ocean waters (Schlesinger & Bernhardt, 2013). Soil carbon content varies dramatically by region and ecosystem type, from agricultural soils that contain an average of only 0.5%–2 % carbon, to peat soils that can be more than 50% carbon (Immirzi et al., 1992; Lal et al., 1995). Wetlands are one of the most effective carbon sinks; with coastal wetlands (which hold so-called "blue carbon") and peatlands that collectively store about 30% of the world's total soil carbon (Lal, 2008; Mitsch & Gosselink, 2007). The rate of annual carbon burial for salt marshes (87.2 ± 9.6 teragrams of carbon per year) exceeds that of tropical rainforests (53 ± 9.6 teragrams of carbon per year), despite their much smaller aerial extent (Figure 2.18, Mcleod et al., 2011).

Despite widespread wetland losses, particularly in North America, these ecosystems play a critical carbon capture role. For example, the soil carbon held in wetlands in the conterminous USA is estimated at 11.5 Pg C, or nearly 1% of the world's total soil carbon (Nahlik & Fennessy, 2016). Canada is home to 25% of the world's wetlands despite losing 32 hectares of wetland area each day, which results in a carbon release equivalent to putting an additional 2,247 cars on the road each day (NAWCC, 2017). Thawing permafrost due to global warming increases microbial decomposition of previously frozen organic carbon, releasing carbon dioxide to the atmosphere, causing a significant positive feedback process (Schuur et al., 2008). For Canadian permafrost, Tarnocai (2006) estimated that 48 Pg C could be released within the current century if the mean annual air temperature increased by 4°C. Wetland protection is critical to prevent further loss of habitat and carbon release. Consequently, Canada has begun to use wetland protection to meet its international greenhouse gas emission targets.

Figure 2 18 **Average long-term rates of carbon sequestration in soils of terrestrial forests compared to coastal wetlands. Note log scale on y-axis. Source: McLeod *et al.* (2011).**



The release of methane from wetlands and other ecosystems offsets some of this natural carbon uptake. Wetlands emit an estimated 115-227 teragrams of methane (CH_4), a potent greenhouse each year, amounting to 20-25% the total global methane emissions. Rates vary by region, for example wetlands in the continental USA emit an estimated 5.5 teragrams per year, while those in Costa Rica produced about 0.80 teragrams per year (Nahlik & Mitsch, 2011), or approximately 0.6% of global tropical wetland emissions. Conversion of wetlands to rice agriculture creates a substantial source. Livestock production also contributes because of cattle's unique digestive tract (enteric fermentation), contributing an estimated 2.2 billion tons of carbon dioxide equivalent greenhouse gases annually (or 35% of total anthropogenic methane emissions). South, Mesoamerica and the Caribbean contribute the most methane (equivalent to almost 1.3 gigatons carbon dioxide) from the production of specialized beef, while North America produces 0.6 gigatons carbon dioxide equivalent (Gerber et al., 2013).

Terrestrial ecosystems affect climate through biophysical feedbacks between vegetation and the atmosphere, and in most circumstances, this is expected to increase the effects of climate change. For example, changing land use alters the surface albedo, or the fraction of solar energy that is reflected from the earth's surface. As tundra snow and ice melt and boreal forests migrate north, highly reflective, white surfaces are replaced by darker vegetation with lower albedo. Areas with darker surfaces absorb more solar energy, leading to higher temperatures. This can increase warming by an additional 1.6°C over the 3.3°C warming predicted if atmospheric carbon dioxide doubles. Similarly, the conversion of tropical forests to pasture replaces forest canopies with pasture grasses, whose leaves are smaller, with lower surface roughness and shallower roots. These traits reduce the cooling effect of evapotranspiration leading to higher local temperatures. At the regional scale, this can reduce annual rainfall and lead to a net warming effect of 1-2°C (Costa & Foley, 2000; Foley et al., 2003).

2.2.10 Regulation of freshwater quantity, flow and timing

There is no substitute for freshwater; it is an essential contribution of nature to people. As water cycles through the biosphere, its distribution varies in ways that determine its utility for domestic consumption (drinking, cleaning), agriculture, industry (including hydropower), transportation, and recreation (Feldman, 2012; Soloman, 2010; Gleick, 2014). Freshwater supply is regulated by terrestrial, wetland, river, floodplain and lake ecosystems. Water is also central in many cultures as a source of identity, livelihoods, as well as a source of customs that inform techniques to use and manage water (Wouters & Tran, 2011).

Freshwater ecosystems are a function of their watershed, or the hydrologically defined land area that integrates the terrestrial areas from which water drains. This includes any stressors that alter water quality and quantity, and the stakeholders that depend directly on the goods and services they supply. The seasonal stability and timing of water supplies are as important as the total annual supply for many domestic, industrial, and navigation uses (Soloman, 2010; Feldman, 2012). Some of this service provision is geophysical, including properties of reservoirs engineered to improve upon natural regulation of freshwater quantity, flow and timing. Vegetation and soils interact with the geophysical of watersheds to intercept rainfall and surface flows, store groundwater, and discharge it more uniformly into surface flows (Brooks et al., 2012). Vegetation and soils interact with the geophysical characteristics of watersheds to intercept rainfall in the canopies (Carlyle-Moses & Gash, 2011), intercept surface flows, store groundwater, and discharge it more uniformly into surface flows (Brooks et al., 2012). Removal of native vegetation as well as afforestation over grasslands or savannas (Jackson et al., 2005) alters the patterns of regional water delivery (Mueller et al., 2013).

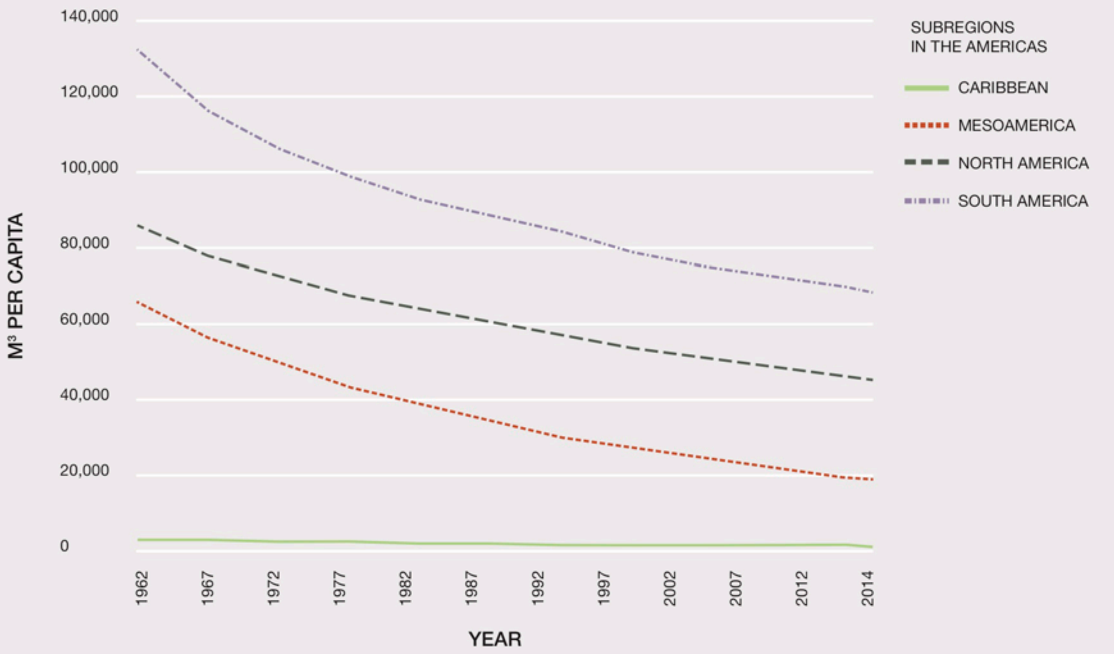
The ecosystems most recognized for the regulation of freshwater supplies are wetlands (MA, 2005; Purkey et al., 1998) and forests (Oswalt & Smith, 2014); including mountain forests in semi-arid to arid regions (Mueller et al., 2013). Wetlands contribute to groundwater storage and the stability of freshwater delivery (Lehner & Doll, 2004; SCBD, 2012). Forests contribute an estimated 53% of human water supplies for the conterminous USA (Oswalt & Smith, 2014). Deforestation decreases evapotranspiration and the interception of rainfall, increasing surface runoff (Foley et al., 2007) and decreasing base flows, such as from the deforested slopes of the tropical Andes (Buytaerti & Breuer, 2013). Throughout the Americas, many wells, human-made water supply impoundments, and water distribution systems have been constructed to increase the reliability of freshwater supply (Cech, 2010).

Freshwater supply has been the subject of economic valuation for a few wetland ecosystems. Values per hectare per year (all values are adjusted to 2016 USA dollars) range from $6 (Troy & Wilson, 2006), to $141 (Roberts & Leitch, 1997), to $8,942. In Brazil, the economic value assigned to the exceptionally large Pantanal wetland is $54/ha/yr (Siedl & Moraes, 2000). Values vary widely as a function of the numbers and distribution of users, scarcity of freshwater supplies, and estimation methods. Market valuation of water supply for various uses may not capture the total value, which can include the nonuse value of scarce biodiversity maintained by part of the supply and various other social-cultural values.

In general, the Americas are rich in freshwater resources, contributing nearly 50% of the total global discharge into the oceans (Fekete et al., 1999). However, water supply varies widely across regions, especially in South America. The supply of freshwater has been subject to increasing pressure as consumptive use, pollution, and populations continue to increase (Postel, 2000; WWAP, 2009; Gleick, 2014), with an average 50% decrease in availability per capita (Figure 2.19). Still, the overall per capita availability is considered to be high in most subregions, except for the Carribean. The availability of renewable freshwater per capita in 2014 was 8,836 m^3 in North America and 22,162 m^3 in Latin America and the Caribbean. However, individual nations vary widely from 315,480 m^3 in Guyana to the lowest supplies in the Caribbean—as low as 282 m^3 in Barbados (World Bank, 2017). Latin Americans use less than 10% of the global total while North Americans use about 15-20%, reflecting the fact that per capita use is over three times as great.

Figure 2 19 Renewable internal freshwater resources in the Americas.

Source: Own representation of data in World Bank (2017). World Development Indicators. https://data.worldbank.org/data-catalog/world-development-indicators. Renewable internal freshwater resources per capita (cubic meters). Last updated: January 3, 2017.



In general, per capita water supply remains sustainable in the Americas, with the exception of the Caribbean, but locally severe water scarcity occurs where high population density intersects with aridity, small river basins and declines in water storage in wetlands and glaciers (e.g. Bogota, Quito, La Paz, Lima; Buytaert & De Biever, 2012). In many high altitude (e.g. Andes) and high latitude regions, glaciers play a significant role in providing water resources for large human populations (Chevalier et al., 2011; Francou et al., 2003). In early 2000 tropical glaciers covered a total area of 1,920 km^2, primarily in the Andes from Colombia to Bolivia, concentrated in Peru (70%) and Bolivia (20%) (Francou & Vincent, 2007; Herzog et al., 2012). Climate warming and deglaciation poses a threat to water supplies for local communities throughout the region, and this is exacerbated by El Niño events (Francou et al., 2003). These circumstances increase reliance on technological solutions for water storage and transport, including foods imported from water rich areas (UN, 2015). For example, infrastructure such as dam building is a common means to stabilize water supplies and regulate flows, although dams also have impacts on other NCP (Palmer, 2010).

However, a trade-off is that technology often adds additional stressors that can impact NCP. Water supply dams, for example, degrade fish habitat services and decrease total water quantity through surface evaporation (Lindstrom & Granit, 2012).

Water supply service are significantly reduced by the conversion of land to agricultural and urban-industrial uses that are less capable of intercepting runoff (Postel & Thompson, 2005). Even where forested ecosystems remain largely intact, past wildfire and livestock management practices can contribute to reduced and more variable total water discharge (Postel & Thomson, 2005; Mueller et al., 2013). Trends show that the conversion of natural ecosystems to agriculture, urban-industrial and other human use is decreasing in the Americas, largely as a consequence of natural areas protection, but significant rates of conversion continue in the Amazon basin and other locations in South America (Soares-Filho et al., 2006), leading to altered precipitation and water supply services.

The restoration of freshwater ecosystems along with improvements in the efficiency of water use (e.g. for agriculture) can reverse many trends associated with impacts to the services they provide (Postel, 2000; Bossio et al., 2009). Large-scale projects in Mesoamerica and North America, for example efforts to restore the Florida Everglades, are designed in part to quantifiably increase various benefits such as groundwater recharge as a source of drinking water for adjacent urban areas. Payment for ecosystems services can incentivize landowners to undertake reforestation and promote water security (NAS, 2016; Lamb et al., 2005).

2.2.11 Regulation of freshwater and coastal water quality

Water of suitable quality is an essential contribution of nature that directly supports human health, high levels of biodiversity, and many types of economic development. High quality water is needed for domestic, agricultural and urban uses, and indirectly contributes to the maintenance of natural fish and shellfish production, water-based recreation, option maintenance, waterborne pest and disease regulation, and other benefits addressed elsewhere in this chapter (Palmer et al., 2009; Layke, 2009; Postel & Thompson, 2005; Mitsch et al., 2001). The capacity of undisturbed terrestrial, riparian, and aquatic ecosystems to regulate water quality is well documented (e.g, Borman & Likens, 1965; Fontescue, 1980; Brauman et al., 2007; Mitsch & Gosselink, 2015; Chapin et al., 2011; Schlesinger & Bernhardt, 2013). Wetlands and riparian zones are particularly effective per unit area (Mitsch & Gosselink, 2015), but upland ecosystems, particularly intact forests and grasslands, are vital because of their larger expanse. Water quality improvement is derived largely from the filtration, retention and sequestration of sediment, nutrients, pathogens, and toxic metals released into the environment by agriculture, industry and mining that, left unchecked, degrade water quality (Starr, 2000; Grigal, 2003; Verhoeven et al., 2005; Sheoran & Sheoran, 2006; Kahn et al., 2009; Ali et al., 2013; Brown & Froemke, 2011).

The benefits of water quality regulatory services have been economically valued for a variety of specific ecosystem and geographical settings, particularly wetlands. As illustrated in Table 2.11, estimates are highly variable, being dependent on environmental and social context, as well as methodology.

Table 2.11. Estimated monetary benefit per hectare provided by the water quality regulatory services in various ecosystem types and locations.

Ecosystem type	Location	Economic Value (US $/ha/yr)	References
Inland Wetlands	North America	1,011 - 2,087	Jenkins *et al.* (2010)
	North America	31,235	Troy &Wilson (2006)
	South America (Brazil)	14	Siedl &Moraes (2000)
Coastal Wetlands	North America	260	Troy & Wilson (2006)
	North America	3,060 - 135,330	Breaux *et al.*. (1995)
	North America	17,840	Costanza *et al.*. (2006)
	North America	19,013	Thibodeau & Ostro (1981)
	Mesoamerica	1,757	Cabrera *et al.* (1998)
	Mesoamerica	28,529	Camacho-Valdez *et al.* (2013)
Forests	Mesoamerica	17	Ammour *et al.* (2000)
	Global Tropical	20 – 1150	Pearce (2001)
	Global Temperate	7 – 68	Pearce (2001)

Degraded water quality is a growing risk to public health, food security and biodiversity (UN Water, 2016). Clean water is a prerequisite to reduce the spread of water borne diseases and vectors that spread disease, such as mosquitoes. Globally, one in nine people do not have access to clean water and more than 3.4 million people die each year from water borne disease (WHO, 2008). These are spread by a variety of species, such as the marsh snail (*Biomphalaria glabrata),* which transmits *Schistosoma mansoni*, and mosquites (*Aedes* spp.) that spread viruses causing, for instance, chikungunya, dengue, and zika. Diarrhea caused by contaminated water and poor sanitation is a leading killer of children and, while declining, accounted in 2015 for 9% of deaths of children under age 5 globally. The Americas have the lowest rates of any region, and by subregion diarrhea accounted for 1% (North America) 5% (Mesoamerica) 2% (Caribbean) and 3% (South America) of deaths for children under age 5, which totals 8,228 for the entire region (WHO & MCEE estimates, 2015). Water pollutants can also contaminate aquatic species used as sources of food, for example the accumulation of toxic compounds in fish tissues in some locations (United Nations Oceans and Law of the Sea, 2016). For example, mercury contamination of fish is reported widely across the Americas, from northern Canada (Scheuhammer et al., 2015), the USA (where fish consumption advisories are in effect in every state; Wentz et al., 2014), and Amazon basin rivers (Webb et al., 2015). Exposure to mercury from fish consumption carries human health risks because it is a neurotoxin (causing damage to the central nervous system at higher concentration) that causes impairments to brain function in children, and acts as an endocrine disrupter at lower concentrations (Wentz et al., 2014).

The variation in water quality between and within subregions is a function of the type, extent, and intensity of land use; how water is used; the degree of economic development, and other stressors (Palmer, 2010). Developing countries typically have less capacity to improve degraded water quality, thus water of substandard quality is often relied on for many uses, including drinking water (Zimmerman et al., 2008). Even if engineered solutions to water quality degradation were available to them, it may not solve all problems. Low enforcement of the law in some developing countries and corruption are responsible for water pollution in basins where industrial activities are prevalent. Some news about Río Santiago in Jalisco, México: http://interactive.fusion.net/river-of-death/

The negative trends in ecosystem regulation of water quality in the Americas are largely due to the conversion of original ecosystems to agricultural and urban-industrial ecosystems maintained for human use. Only recently have we recognized trade-offs between the benefits from ecosystem conversion and lost water quality benefits (Foley et al., 2005; Brown & Froemke, 2012; section 2.7 for a more thorough discussion of trade-offs). Agricultural lands, with characteristically high nutrient runoff, cause widespread eutrophication of inland and coastal waters, as well as hypoxic 'dead zones' throughout the Americas, most famously in the Gulf of Mexico off the shore of the USA (Diaz & Rosenberg, 2008), all of which can degrade commercial fishery and recreational services, influencing culture and livelihoods. The occurence of dead zones has increased exponentially since the 1960s (Diaz & Rosenberg, 2008). Where data on trends are available, water quality is often declining, for example national survyes in the USA streams and rivers show that more than 40% of stream miles suffer from nutrient pollution, and over the period 2004-2009, 9% fewer stream miles were rated as having good overall water quality and high levels of nutrients. Degradation of water quality diminishes its use for human consumption, for instance algal blooms of *Microcystis spp.* and other cyanobacteria can release microcystin, a potent liver toxin that has safeguards set by the World Health Organization. Between 2007 and 2012, an assessment of USA lakes showed a nearly 10% increase in the detection of micocysitn. Extreme events result in beach closings and contaminate potable water supplies, such as the drinking water ban that occurred in Toledo, OH during the summer of 2011 (Paerl & Huisman, 2008; Michalak et al., 2013). Wetland restoration has been suggested as a means of recreating the ecosystem services that reduce nutrient runoff and regulate water quality in highly agricultural watersheds (Mitsch et al., 2001).

Future trends in the Americas are uncertain but as human populations and economies grow, the demand for clean water will increase and could exceed supply by 40 % (UN, 2013). Water quality issues are increasing in some developing areas where rapid urbanization and industrialization are responsible for acute water pollution problems. The Río Santiago in México, which has toxic levels of arsenic from industrial waste, is an example (Rizo-Decelis and Andreo 2015; Fusion, 2015 http://interactive.fusion.net/river-of-death/). Engineering solutions to water quality problems have been effective in the past, but are expensive, rely on fossil fuels for power, and are impractical for dispersed (non-point) sources of pollutants in agricultural and urban ecosystems.

2.2.12 Regulation of hazards and extreme events

People are periodically exposed to hazardous and extreme events that diminish their quality of life (Smith, 2013; Shi & Kasperson, 2015). Nature often contributes to the moderation of extreme events that include floods, storm damage and storm surges, landslides (including avalanches), droughts, extreme heat, windstorms, and fire.

In river flooding, the peak discharges of streams and rivers are moderated by the capacity of watersheds to divert water into surface and groundwater storage (Dunne & Leopold, 1978; Bosch & Hewlett, 1982; Deberry, 2004; Brooks et al., 2012). Among the better predictors of peak stream discharge are watershed slope, soil saturation, and the amount of impervious surface, either natural or human made. The presence of plant and animal communities typically increase surface roughness, which slows water flows and increases infiltration into short- and long-term groundwater storage (Brooks et al., 2012). In mountainous areas, evergreen trees help hold the snow in place and shade it from rapid melting (Bosch & Hewlett, 1982; Harr, 1986).

Vegetation also moderates the chances of snow avalanches and landslides that are caused by events such as earthquakes and extreme precipitation events. While the slope and underlying geological structure are major determinants of the size and extent of avalanches and landslides (Lu & Godt, 2013; Ren, 2015), vegetation can have moderating effects on surface structure, such as the ability of tree roots to bind slope substrates into forms more resistant to slope slippage. For instance, in the Andes, the likelihood of landslides increases with land use and time since deforestation (Vanacker et al., 2003).

Extreme heat, drought, and fire are typically viewed as threats to ecosystems (Daily, 1997; Allen & Breshears, 1998; Sun et al., 2015), but many ecosystems have evolved with natural drought, heat, and fire and some natural ecosystems can moderate local drought, heat, and fire effects, largely through their influence on water storage capacity (Brooks et al., 2012), shade and transpiration (Jenerette et al., 2011). Climate regulation at global scales is addressed in section 2.2.9. The vulnerability of forest ecosystems to fire and other sources of stress increases when they are stressed by disease, heat or water deficiencies (Barnes et al., 1998), or by poor management (Omi, 2005). The costs of wildfire damage and management are increasing (Gorte, 2013). Part of this cost appears to be due to degraded natural services resulting from poor management practices (Omi, 2005). Improved management includes proactive actions such as prescribe burns and managed buffers between forests and residential areas, but is costly. Reactive wildfire management costs alone are high, in part because of insufficient investment in proactive management. For example, the Federal appropriations for fighting wildfires in the USA was nearly $3 billion in 2012 and has in general been increasing as the size and frequency of fires has increased in response to environmental changes (Gorte, 2013).

While storms (including hurricanes) have significant effects on coastal ecosystems (Lugo, 2008; Mitchell, 2012; Morton & Barras, 2011), coastal wetlands and coral reefs moderate hurricane impacts on coastal communities, buffering against storms and storm surges (Costanza et al., 2008; Bravo de Gueni et al., 2009; Barbier & Enchelmeyer, 2014; McIvor et al., 2012; Van Zanten et al., 2014). For example, existing coastal wetlands reduced damage from "Superstorm Sandy," which hit the USA east coast in 2012, by an estimated $625 million. In response, shoreline modification in the New York City region now includes restoring salt marsh habitat as an alternative or accompaniment to 'hard' infrastructure (Grime et al., 2016). Although studies are few, mangrove ecosystems can moderate storm surges by slowing water flow and reducing wave action, with an estimated 5 to 50 cm decrease in water levels per kilometer width of mangroves (McIvor et al., 2012).

Anticipated climate change could increase the impacts of hazardous events in various ways (IPCC, 2007), placing more stress on ecosystems and more pressure on whatever mitigating services they may provide. The impacts of climate change, however, may be moderated by reducing local human impacts on ecosystems. Recent trends indicate that more ocean, land, and wetland areas are now protected than in the past (section 2.2.8).

Nature can improve quality of life by providing necessary resources and space to recover after extreme events. For example, in a study conducted in Valdivia, Chile, urban wetlands were found to be one of the most mentioned urban spaces that were used for earthquake recovery. However the actual use of those spaces vary depending on their biophysical characteristics that modify their utilitarian benefits and therefore the level of protection they provide (Barbosa & Villagra, 2015). Other examples are places such as plazas, parks and free areas, which after catastrophes, are used as places for refuge and can potentially satisfy the need for adaptation (Villagra et al., 2014). This is consistent with the services that the use of green spaces offers. As recognition of the role of natural ecosystem functions has increased, they are increasingly included in what is called "green infrastructure" or "nature-based infrastructure (Niemela, 2011) as management measures (Benedict & McMahon, 2002; Cunniff & Schwartz, 2015).

2.2.13 Habitat creation and maintenance

In landscapes dominated by anthropogenic land use, such as agriculture and silviculture, but also in cities, the presence of natural habitat in sufficient amount is of high importance both for biodiversity maintenance and for humans. In Buenos Aires, Argentina, for example, a study recognizes the importance of green spaces as places of opportunity for education and engagement with nature (Morello & Rodriguez, 2001). In Colombia, the Medellin Green Belt project´s aim is to "create a healthier urban living environment for humans and nature alike" and eventually devote this area for landscape restoration to better support native biodiversity (Pauchard & Barbosa, 2013). These initiatives in the southern hemisphere are important, as urban vegetation has not nearly achieved the same attention as it has in northern hemisphere cities (see Niemelä et al., 2011). In urban areas, green spaces underpin ecological functions that result in NCP to society (Barbosa et al., 2007), which corresponds to the concept of green infrastructure that not only includes natural vegetation or green spaces in general, but also human modified green structures such as green walls and roofs, eco-bridges and corridors, artificial wetlands etc., all of which provide some benefits for biodiversity or humans, especially in, but not restricted to, cities.

In agriculture ecosystems, the creation of habitat for biodiversity maintenance has been related to several benefits to people directly (habitat for fisheries, for game species, medicinal plants, water quality improvement and to prevent soil loss), and indirectly by benefiting their crops or other production systems (biological control, pollination, dung burial by beetles; e.g. Steel et al., 2017; IPBES, 2016; Weyland & Zaccagnini, 2008; Viers et al., 2013). A variety of ecosystem types such as natural and created wetlands, riparian areas, hedgerows, vegetation strips, and vegetation islands placed between continuously cropped areas, serve as corridors for the movement of different species groups increasing biodiversity locally

(Goijman & Zaccagnini, 2008; Zaccagnini et al., 2014) and regionaly (Goijman et al., 2015). They also contribute ecosystem services by improving downstream water quality by filtering agricultural chemicals (Peterjohn & Correll, 1984; Hilty & Merenlender, 2004; Fennessy & Craft, 2011). Ecosystem services such as pollination, can increases with a proper landscape design such as interspersing crops with wild lands and native habitat patches (Brosi et al., 2008). For example, coffee yields increased fully 20% in Costa Rica as distance between fields and native forests decreased (Ricketts et al., 2004).

Legislation that aims at maintaining natural vegetation within agricultural landscapes is of high relevance, for example in Brazil, where by law at least 20% of natural ecosystems (80% in forest area and 35% in savanna areas in the Legal Amazon Region; Federal law 12.651 from May 25th, 2012) must be maintained in any rural property above a certain size in the so-called Legal Reserve, and where ecosystems adjacent to rivers, on steep hillslopes and hill tops are placed in Permanent Protection Areas. However, it is important to consider the scale (extent and distances) of natural elements in the landscape: optimum values will depend on the benefits to be achieved. Landscape heterogeneity and multifunctionality usually provide most habitat functions and several other benefits (Landis, 2017).

Nonetheless, throughout the Americas, natural ecosystems have been widely destroyed, mostly for production of food or other benefits. An example is Brazil's Atlantic forest region, one of the regions that first were subjected to dramatic land use change in the Americas. Here, today only 11-16% of area are covered by natural ecosystems remain, and most of them small and fragmented (80% of area in patches of less than 50 hectares; Ribeiro et al., 2011). In many other countries, some regions have seen similarly strong land use change and thus losses in biodiversity and ecosystem services. Ecological restoration has been recognized as critical to maintain or recover biodiversity, NCP, and human wellbeing (Aronson et al., 2006; Perring et al., 2015), and ambitious goals have been established throughout the world. A prominent example is the Bonn Challenge, where more than 30 countries, including 13 from the Americas, committed to restore 150 million hectares of the world's deforested and degraded lands by 2020 (including 44.9 million hectares in the Americas), and 350 million hectares by 2030 (Table 2.12; www.bonnchallenge.org). The importance of these efforts is highlighted by the fact that the Americas house one-third (or eight) of the originally proposed 25 biodiversity hotpots (Myers et al., 2000), including in the Caribbean and Mesoamerica. Recovery of NCP means that restoration goals must go beyond the restoration of biodiversity and consider ecological processes and services, as well as social and economic aspects (e.g. Wortley et al., 2013; Kollmann et al., 2016). Where possible, care must be taken to 'restore' an ecosystem that is based on the characteristics of the original one (see Veldman et al., 2015). However, it may not always be possible to restore original conditions, especially as global climate change shifts habitat conditions and species distribution ranges. In the context of climate change, the maintenance or restoration of reasonable amounts of natural habitats and of corridors that connect them is critical to promote the adaptation of natural ecosystems to climate conditions: ecological corridors are critical for dispersal processes in the landscape and the migration of species in reaction to human activities (Robillard et al., 2015; Haddad et al., 2000). For example, these serve as stepping stones for migrating species through California's agricultural landscapes (Hilty & Merenlender, 2004).

Table 2.12. Restoration commitments of countries from North America, Mesoamerica and South America to the Bonn Challenge (no Caribbean countries have made commitments).

Country	Bonn Challenge Commitment (hectares)
North America	
USA	15 million
Mesoamerica	
Costa Rica	1 million
El Salvador	1 million
Guatemala	1.2 million
Honduras	1 million
Mexico	6.5 million
Panama	1 million
South America	
Brazil	12 million
Peru	3.2 million
Argentina	1 million
Colombia	1 million
Chile	0.5 million
Ecuador	0.5 million
Source: www.bonnchallenge.org	

A top priority is to protect and maintain natural habitats in agricultural landscapes (Scherr et al., 2008). Such 'ecoagricultural' landscapes simultaneously provide multiple benefits of nature to people, including food production, biodiversity support, with less environmental impact. Here, organic farming also makes important contribution. Organic farming has increased considerably in the past years throughout the Americas, with Argentina (3,073,412 ha), the USA (2,029,327 ha), Uruguay (1,307,421 ha), Canada (944,558 ha), and Brazil (750,000 ha) leading as countries with the most area under organic farming practices (Willer & Lernoud, 2017). Still, this is only a small fraction of total agricultural area in most countries in the Americas.

Before intensive agriculture was spread globally, fueled by the green revolution, agricultural activity relied entirely on ecosystem services such as soil formation and fertilization, natural pest control. Some of these natural ecosystem functions have diminished or disappeared from intensive crop fields – and been replaced by chemical (e.g. pesticides, fertilizers) and energy inputs (combustibles).

Importantly, indigenous and local management practices can contribute to enhance NCP both in terrestrial and aquatic ecosystems. For instance, Begossi (1998) describe how the diversity of management practices regarding small-scale slash-and-burn agriculture and fisheries produce NCP and increase resilience in local communities of Cablocos from the Amazon rain forest and Caiçaras from the Atlantic forest in Brazil. Another example is the many First Nations groups in Canada restoring and/or enhancing stream habitats for salmon fisheries (Garner & Parfitt, 2006).

2.2.14 Regulation of air quality

Ecosystems have an important role in regulating air quality through the exchange of trace gasses and deposition of particulate matter. This can have positive effects on air quality as pollutants are removed by interception by, or deposition on vegetation. The deposition of nutrients (e.g. nitrogen) in moderate amounts can increase primary productivity, particularly in areas where nitrogen is limiting (Dise et al., 2011). However, excessive deposition may damage vegetation, reducing its capacity to provide this and other benefits to human well-being. In some cases vegetation can have a negative effect by emitting precursors to other, more serious air pollutants.

Human activities related to industry, energy generation, and transportation generate emissions that diminish air quality by releasing particulate matter, nitrogen oxides, ammonia, sulfur dioxide, and carbon monoxide (Smith et al., 2013). The costs of particulate and gaseous air pollutants to human health can be considerable, although these are not well quantified in many subregions of the Americas. Outdoor air pollution is a major environmental health risk, particularly in urban areas where sources of pollutants are concentrated. Fine particulate matter (<2.5 µm, or particulate matter2.5) is strongly linked to diseases such as lung cancer, and pulmonary and cardiovascular diseases. In 2014, an estimated 90% of people globally living in cities experienced particulate matter at levels above World Health Organization guidelines. Limited progress in improving air quality over the past decade points to the difficulty in reaching the Sustainable Development Goal 11.6, related to reducing the adverse environmental impacts of living in cities, including those related to air quality (WHO, 2016). In regions where air pollution is high, other services such as crop production and those related to forest growth (carbon sequestration, support of biodiversity) can be impacted (Grimm et al., 2008; Dise et al., 2011). Globally it is recognized that production of air pollutants in one region (e.g. from industrial activities or forest/biomass burning) can circulate to other regions contributing to negative human health effects and crop damage (Akimoto, 2003; Hollaway et al., 2012). For example, nitrogen oxides emissions from North America lead to ozone formation and crop production losses, particularly to corn and soybean) in Europe and other portions of the northern hemisphere (Hollaway et al., 2012).

Urban forests and street trees are increasingly recognized as contributing to improved air quality with associated reductions in health risks. The ability of trees to absorb pollutants and promote deposition of particulates can directly benefit human health, although much of the evidence is through modeling estimates at regional scales, making site specific predictions difficult (Salmond et al., 2016). The benefits that trees and other vegetation provide have resulted in programs to promote tree planting and the 'greening' of cities (Salmond et al., 2016). The demonstrated value of urban trees includes cooling of local temperatures and mitigation of heat stress, both by shade and evaporative cooling. The removal or particulates from air provides substantial benefits. Modeling studies show that urban forests in Santiago, Chile remove an estimated 14.8 – 17.3 g particulate matter10 per m^2 per year, effectively increasing air quality (Escobedo et al., 2008). Parks can also have substantial benefits, for example, vegetation in a peri-urban park in Mexico City, one of the most air polluted cities in the Americas, reduced ozone by 1%, particulate matter10 by 2%, and carbon monoxide by 0.2% of the annual concentration (Baumgardner et al., 2012). In a recent review, 89% (34 of 38) of studies examining air quality showed a demonstrated improvement due to the reductions in particulate matter, ozone, sulfur dioxide, nitrogen dioxide, and carbon monoxide (Roy et al., 2012). Economically, tree planting in urban areas has been reported to have a net benefit, with cost benefit ratios of 3.8:1 to 4.5:0 (Salmond et al., 2016).

Some ecosystems are sources of air pollutants (disservices), for example agricultural systems that emit ammonia and nitrite as a result of fertilizer use and livestock operations. A disservice of urban trees is the emission of gasses that are precursors to secondary pollutants for example, volatile organic compounds that are involved in the formation of ground level ozone (so called "bad ozone" to distinguish it from good ozone in the upper atmosphere that blocks harmful UV radiation) (Horowitz, 2006; Salmond et al., 2016). The emission of these compounds has been shown to vary by tree species (Roy et al., 2012). Increasing ground level ozone has been linked to reduced lung function and worsening of existing conditions such as emphysema (WHO, 2005). Ozone pollution has also been documented to reduce crop production through damage to staple crops such as soybean, maize, and wheat. In 2000, global reductions in yield due to ozone exposure were estimated as 8.5-14% for soybean, 3.9-15% for wheat, and 2.2-5.5% for maize, worth an estimated $11-18 billion (Avnery et al., 2011).

Finally, trees and other vegetation, particularly in urban areas, also provide important social and cultural values by providing opportunities for recreation, aesthetic enjoyment, and allowing residents an opportunity to 'connect with nature' (Roy et al., 2012).

2.2.15 Regulation of organisms detrimental to humans

Human health is intimately interconnected with biodiversity and the health of our ecosystems. There are different ways in which biodiversity can provide health and well-being to humans, thus improving quality of life, including psychological, physiological (e.g. food provision), and traditional and modern medicines. Another important benefit from biodiversity to human health is the capacity to regulate the transmission and prevalence of some infectious diseases.

The causes behind disease emergence in humans are similar to those affecting the loss of biodiversity, including habitat change, overexploitation and destructive harvest, pollution, invasive alien species and climate change (Romanelli et al., 2015). In particular, the connections between animals and environment and the emergence of infectious diseases in humans are highly relevant (Taylor, 2001). For example, the majority of human infectious diseases emerged from zoonotic pathogens (transmitted from animals), with most of these caused by pathogens with a wildlife origin, including the emergence of HIV/AIDS (Human Immunodeficiency Virus Infection / Acquired Immune Deficiency Syndrome; from primates hunted for human consumption), which has caused millions of human deaths as well as an economic and health burden for the past 40 years (Jones et al., 200; Allen et al., 2017; Ostfeld, 2017).

Land use changes driven by road building, deforestation and expansion of agricultural fields are a main cause of outbreaks of infectious diseases, including the emergence of new pathogens (Loh et al., 2015; Romanelli et al., 2015). Documented examples exist for increased transmission of Dengue fever, yellow fever, leishmaniasis (Walsh, 1993; Willcox & Ellis, 2006) and malaria (Walsh, 1993; Vittor et al., 2006; Pattanayak & Yasuoka, 2008). Models that link malaria epidemiology with socio-economic and demographic data shows an increase in prevalence in early stages of land development, followed by a decrease in cases over time (Baeza et al., 2017). Depending on the type of land cover and socio-economic factors, land use change can lead to a higher or lower rates of malaria transmission compared to undisturbed areas. Mining activities and hydroelectric dam building have been shown to be reservoirs for malaria (Bardach et al., 2015; Castellanos, 2016). In general, vector-borne disease incidence is also likely to increase as hydroelectric dams proliferate on the Amazon and its tributaries, even as some consider hydropower a clean energy source. In regions with large hydropower plants, the rate of malaria is 278 times higher than in forested areas (Afrane et al., 2006).

Increased harvest and exploitative practices, such as hunting and mixing of wildlife and domestic species in markets, can also change the pathogen dynamics and favor the spillover and further spread of pathogens in humans. For example, in 2009 an outbreak of influenza-like respiratory illness started in Mexico and quickly spread through the world. When the pathogen responsible for this outbreak (H1N1 virus) was isolated, the genetic composition showed a reassortment of genes from a variety of domestic, wildlife and human origin, including the North American and Eurasian avian virus, human virus and swine virus (Neumann et al., 2009). One month after the initial outbreak 41 countries reported more than 11,034 cases, including 85 deaths (Novel Swine-Origin Influenza A (H1N1) ((Novel Swine-Origin Influenza A (H1N1) Virus Investigation Team, 2009). The economic impact to Mexico’s tourism industry totaled nearly $3 billion in losses plus a pork trade deficit in the tens of millions of USA dollars (Rassy, 2013).

Environmental pollution poses direct threats to biodiversity and human health. In particular, the use of antimicrobials for humans and animal medicine as well as food production can disrupt microbial composition and also can lead to develop anti-microbial-resistant infections. Similarly, contaminated water could enable the long-term persistence of human pathogens such as *Vibrio cholerae*, leptospirosis and parasitic worm-transmitted schistosomiasis, and may promote growth of harmful algal blooms that may be toxic to marine life (including food sources) and even directly to humans.

Biodiversity and human health are also likely to be affected by climate change and extreme weather events. For example, shifts in species ranges may facilitate the redistribution of hosts and their pathogens (Pecl et al., 2017). Future forecasts of precipitation and temperature suggests that mosquito vectors (e.g. *Anopheles*, *Aedes*) will reach new suitable areas with the poleward and elevation migrations, particularly in tropical regions (Siraj et al., 2014). For example, climate change may play an import role shaping the suitability for vector-borne diseases such as malaria (Caminade et al., 2014). Human populations may also suffer health impacts from extreme weather (e.g. heat or cold exposure injuries and water-borne diseases from flood events).

Alteration of species diversity dynamics, particularly community composition can potentially affect infectious disease transmission (Terborgh et al., 2001; Ostfeld & Holt, 2004; Rocha et al., 2013) and have further negative effects on humans. For example, increased acai plantations and removal of wildlife in the Brazilian Amazon has led to a higher number of Chagas disease cases in the region (Araújo et al., 2009; da Xavier et al., 2012). By feeding on already ill or disabled individuals, predators may also play an important role in controlling the emergence and spread of diseases. Changes to species migration (e.g. via habitat fragmentation) can displace wild animal populations, and may create negative novel species interactions, particularly around forest edges.

In addition, other studies have proposed the dilution effect hypothesis stating that high biodiversity could reduce the risk of pathogen transmission (Norman et al., 1999, Ostfeld & Keesing, 2000; Johnson & Thieltges, 2010). This pattern has been observed in different disease systems such as Hantavirus (Suzan et al., 2009), Lyme disease (Werden et al., 2014), West Nile virus (Allan et al., 2009) among others (a detailed review can be found in Ostfeld & Keesing, 2012). Several studies have also contested the generality of the dilution effect and consider that it only applies under specific circumstances (Salkeld et al., 2013; Wood et al., 2014). In general, studies should take into account the particular host-pathogen system, the scale of analysis and the risk indicator used (Huang et al., 2016).

Even non-zoonotic disease may have indirect impacts on human health and well-being. For example, declines in bats due to disease like white nose syndrome in North America could affect the production of the ecosystem services they provide – among them, pest control and pollination (Boyles et al., 2011). Avian scavengers, such as vultures ,provide an essential ecosystem service through their scavenging on carrion, preventing pathogen contamination of water bodies and food sources. However, certain chemicals, including some insecticides and rodenticides, can be highly poisonous to scavengers. The scarcity of research on scavenger exposures to toxins (e.g. lead) in Latin America has been noted, which is particularly concerning given the continued use of some potent pesticides in South America (Lambertucci et al., 2011).

In some cases, wildlife may serve as sentinels for human disease risk. In 2012, a report of six howler monkey carcasses found near a wildlife sanctuary in Bolivia led to rapid specimen collection and screening. The Ministry of Public Health was notified upon detection of a Flavivirus, and preventive vaccination and public health awareness campaigns were launched. Further testing ultimately indicated infection with Yellow Fever virus – the first such mortality event in howler monkeys ever reported in the country. No human cases were reported, likely due to the swift information sharing and mobilization of prevention measures (Uhart et al., 2013).

2.2.16 Pollination and dispersal of seeds and other propagules

Pollination is an ecosystem function that is fundamental to plant reproduction, food production and the maintenance of terrestrial biodiversity. As an ecosystem service, more than 75% of the leading types of global food crops benefitting from animal pollination (IPBES, 2016). As a result pollination is also important to social and economic systems that directly affect human well-being. For example, this NCP represents billions of USA dollars annually to local and national economies (Table 2.13).

Table 2.13. Economic valuation of nature contributions to people via pollinators and pollination in the Americas.

Subregion	Economic contribution of pollination NCP
North America	It has been estimated that insect-pollinated crops directly contributed $20 billion to the USA economy in the year 2000. If this calculation were to include indirect products, such as milk and beef from cattle fed on alfalfa, the value of pollinators to agricultural production would be raised to $40 billion in the USA alone (Marks, 2005).
Mesoamerica	In Mexico, the overall income generated by non-pollinator-dependent crops is considerable smaller amount compared to that obtained from pollinator dependent crops which represents 54% of the yield value, it means, in terms of productivity, pollinator-dependent crops produce significantly more volume (Ashworth *et al.,* 2009)
South America	Giannini *et al.* (2015) estimated the economic value of pollination for 44 crops in Brazil and found that the highest values obtained were for soybean (~$5.7 billion, DR=0.25), coffee ($1.9 billion, DR=.25), tomato ($992 million, DR=0.65), cotton ($827 million, DR=0.25), cocoa beans ($533 million, DR=0.95), and orange ($522 million, DR=0.25). The total value of annual production of dependent crops was $45 billion, and the total contribution of pollinators corresponded to $12 billion, that is, 30% of the total production.
	Pollination provided by wild bees is a biodiversity-linked ecosystem service that is likely to common in the Andean montane environment. Biotic pollination is common at all latitudes and altitudes of the Andes (Arroyo *et al*., 1982, Kessler, 2001b, Aizen *et al.,* 2002, Kay *et al*., 2005, Kromer *et al.,* 2006, Barrios *et al*., 2010, Smith-Ramirez *et al.,* 2014).
	In Colombian coffee plantations, the value that could be lost in farmers' income from a reduction in pollinators for native bee pollination (i.e. stingless bees) was calculated to be $16.5 ± 33.2 per hectare per 2010/2011 harvest (1.7 ± 0.8% of farmer's net revenue), and $129.6 ± 65.7 per hectare per 2010/2011 harvest (3.7 ± 0.9% of farmer's net revenue) for honeybees. The large difference in valuation between stingless and honeybee values is noteworthy and the narrow range of variability for stingless bees (Bravo-Monroy *et al.,* 2015).
Dependence rate on pollinators (DR) is classified as being: essential, DR=0.95 (meaning that the value of pollination-driven yield lies between 90 and 100%); great, DR=0.65 (40–90% of yield is dependent on pollination); and modest, DR=0.25 (10–40% of yield is dependent on pollination).	

Plants in the Americas are pollinated by several wild species including native bees and bumblebees, butterflies, moths, wasps, beetles, birds, bats and other vertebrates. Crops are mainly pollinated by introduced honey bees (*Apis mellifera*) and bumblebees (e.g. *Bombus terrestris*) (Committee on the Status of Pollinators, 2007). Although bees are considered to be the most important pollinators, insects other than bees are efficient pollinators as well and provide 39% of visits to crops (Rader et al., 2015). There are also local products in which pollinators play a key role; for instance, the *kapok* is a bat-pollinated tree that produces silky fibers used in bedding and cushion materials and also many bat-pollinated cacti throughout the Americas produce edible fruits (Garibaldi & Muchhala, 2011). In addition, seagrasses that form extensive meadows in shallow marine waters are pollinated by invertebrate fauna (van Tussenbroek et al., 2016). These seagrasses are amongst the world's most productive ecosystems and provide several NCP, such as habitat maintenance, regulation of freshwater and coastal water quality and protection and decontamination of soils and sediments.

Indigenous and local knowledge of native bee species included specific emphasis on stingless bees (Table 2.14). Today, managed pollination is largely based on *A. mellifera*, an exotic species for America, which has become the major commercial pollinator, as well as other bee species and bumblebees. Displacement of native pollinators by *A. mellifera* has also occurred in Mexico (Pinkus-Rendon et al., 2005) and by *B.*

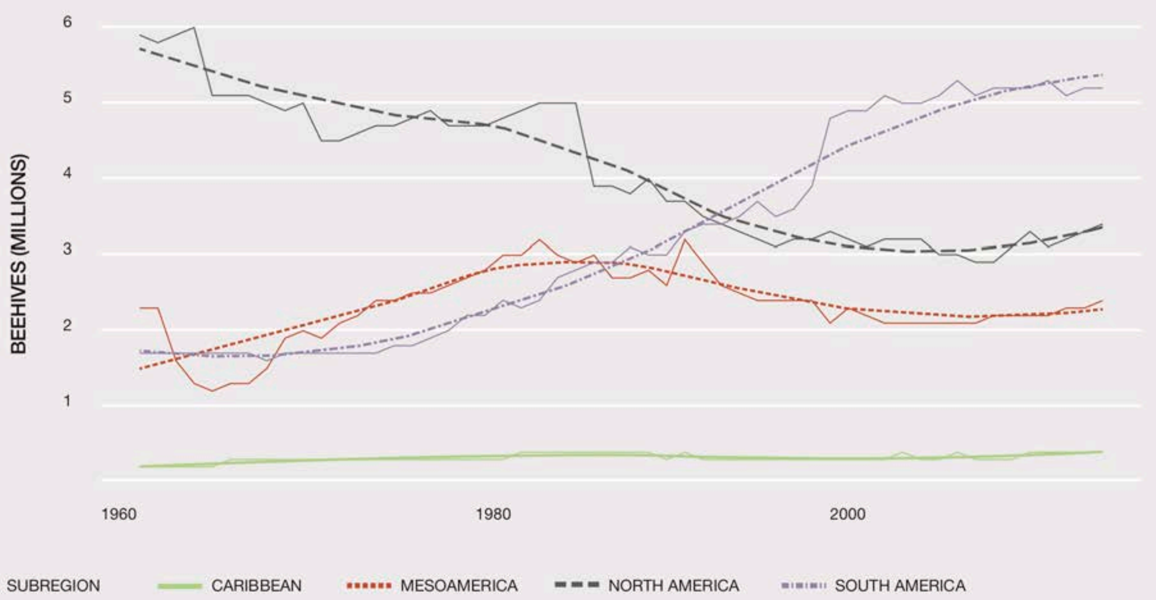
terrestris in Chile and Argentina (Morales & Aizen, 2008). In the Americas region, the number of managed colonies in the last 50 years has increased from 10 to 11.3 million beehives in 2014; in South America the number of colonies has increased, but in North America this trend has decreased (Figure 2.20). As a reduction in colony numbers will lead to a reduction in pollination services, this relationship is not fixed. For example, in the USA beekeepers face trade-offs between the quantities of honey they can produce and the earnings they get from pollination services, since the movement of colonies places stress on the bees and reduces honey productivity (Burgett et al., 2010). Honeybees play a key role to increase yields and the quality of many crops for food production, considering that 50% of the cultivated area relies on pollination, around 1.5 million of colonies are needed to satisfy the global demand (Pirk et al., 2017). The importance of bees for food production (cereals, vegetables, fruits and honey) has stimulated a detailed indigenous knowledge in this pollinator (Box 2.5.).

Table 2.14. Bee species diversity links with indigenous and local knowledge of stingless bee pollinators.

Country	Total Bee Species	Stingless Bees # (%)	Stingless Bees Used # (%)
Mexico	1,795	46 (2.6)	19 (41.3)
Costa Rica	785	58 (7.3)	2 (4.2)
Colombia	541	101 (20.0)	17 (16.8)
French Guiana	210	80 (38)	2 (2.5)
Peru	688	100 (14.5)	12 (12)
Brazil	1,814	236 (13.0)	21 (8.9)
Source: Ayala *et al.* (2013).			

Figure 2 20 Number of managed colonies of beehives by subregion, 1961–2014.

Note: The stat_smooth function was applied in R (ggplot2 package) to get the smooth lines.Source: FAO (2017). FAOSTAT Statistics Database. http://www.fao.org/faostat/en/#data/QA. Date accessed: April 11, 2017.



Wild pollinators have declined in occurrence and diversity at local and regional scales. Some of the drivers of pollinator decline are: i) land-use change; ii) intensive agricultural management and pesticide use; iii) environment pollution; iv) introduction of alien species: plants, pollinators, pests and pathogens and v) climate change (IPBES, 2016; Potts et al., 2010). Recent studies suggest that viruses found in (*A. mellifera*) have recently been detected in other wild bee species (Tehel et al., 2016) and has the potential to make that population decline in those species. The predicted climate change may affect negatively several species associated with tomato production in Brazil (Elias et al., 2017).

Box 2.5. The importance of indigenous and local knowledge of bee pollinators

According to Jones (2013), in Argentina one of the first travelers from Europa wrote: "An Indian goes into a wood with an axe and the first tree he comes to that has an entrance hole to a bees' nest. By boring other holes he gets five or six jugs of pure honey. These bees are small and have no sting…" Breeding and handling of the stingless bee *Melipona beecheii*, also known as *xunan kab*, is the longest traditionally managed bee in Mesoamerica (Villanueva-Gutierrez et al., 2013). The practice of beekeeping by ancient Mayans was documented by one of the Mayan codices (the *Tro-Cortesianus* codex), dating from the Postclassic period of Mesoamerican chronology (*circa* 900–1521 CE), and it is estimated that there are 46 stingless bees species in the Mayan territory (Lyver et al., 2015). In Brazil, the Kayapó also breed stingless bees, using the honey for both daily and ritual uses. Studies have also registered the knowledge of the Guarani and Pankararé tribes related to morphologic and ethological descriptions, distribution, and dispersal of bees, as well as practical issues related to manipulation and extraction of honey. The Enawene-Nawe group recognized 48 stingless bee species, and this knowledge even helps clarify the biology of some species (dos Santos & Antonini, 2008). In Costa Rica, there are 20 stingless bee genera and 58 species present, and 20 different hived or semi-domesticated species have been reported in the provinces of Guanacaste, Puntarenas, San José, Cartago and Heredia (Vit et al., 2013). In summary, stingless bees are economically, ecologically and culturally important to many indigenous peoples and local communities in the Americas. They are one of the most important pollinators of native and cultivated tropical plants, while products such as honey, pollen and cerumen have also been used by indigenous and non-indigenous people in the Americas (Ayala et al., 2013).

2.2.17 Regulation of ocean acidification

One fourth of the carbon dioxide released into the atmosphere from anthropogenic activities is absorbed by the ocean (Le Quéré et al., 2010), and since the industrial revolution about 375 billion tons of carbon have been emitted to the atmosphere as carbon dioxide (WMO, 2012). When carbon dioxide enters the ocean it changes seawater chemistry, resulting in increased seawater acidity; the ocean has become 27% more acidic since the beginning of the industrial revolution. Increased acidification reduces the concentration of calcium carbonate ($CaCO_3$), which poses a major threat to calcifying marine organisms, such as coral (Raven at al., 2005; Kleypas et al., 2006). Yields from commercial and subsistence fisheries are expected to be reduced substantially, especially for shellfish fisheries (Cooley & Doney, 2009), although the magnitude of reductions depends on many social and economic aspects of the fisheries and their capacities to adapt (Voss et al., 2015). Both coastal warm-water and deep-sea cold water coral reefs, are biodiversity hotspots (see Chapter 3), and also seriously threatened by increasing ocean acidity (Mora et al., 2016), with again limits to capacity for adaptation ecologically (Khan et al., 2015) and for the communities dependent on coral reefs for livelihoods.

Ocean acidification is affecting not just marine biota directly dependent on $CaCO_3$ for physical structure, but also the people that depend on the marine biota for livelihoods and food security. However, coastal ecosystems can help to address this threat from climate change. Coastal blue carbon ecosystems (mangroves, tidal marshes, and seagrasses) represent important climate mitigation opportunities due to their ability to function as carbon sinks, sequestering carbon dioxide from the atmosphere and oceans (Chmura et al., 2013; Lavery et al., 2013). For instance, vegetated wetlands occupy only 2% of seabed area, yet represent 50% of carbon transfer from oceans to sediments (World Bank et al., 2010). Evidence is emerging that suggests mangroves may be able to partially mitigate acidification of coastal tropical waters (Sippo et al., 2016).

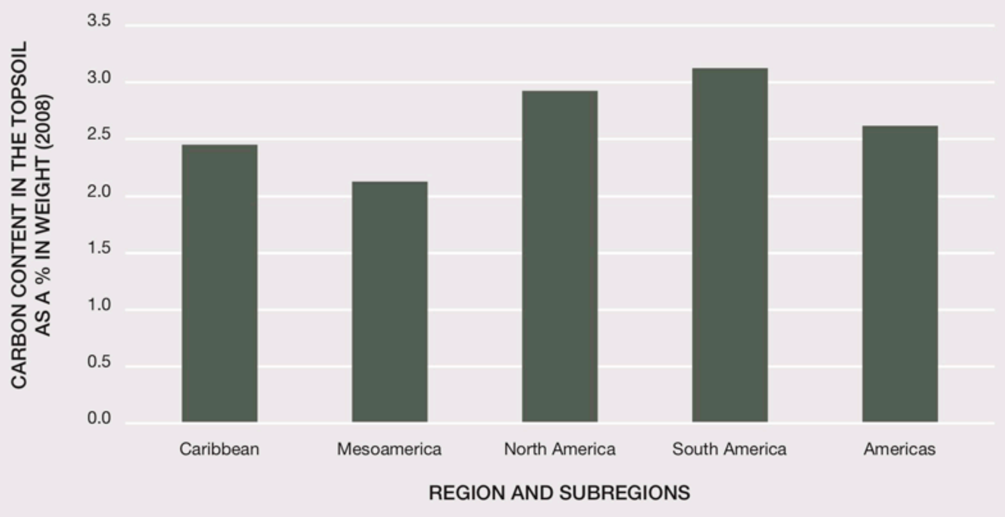
2.2.18 Formation, protection and decontamination of soils and sediments

Soil is a multiphase system composed of solids (minerals, organic matter and biota), liquids and gases (Ugolini & Spaltenstein, 1992). It is a source of water and nutrients for plants and microorganisms and is the physical support systsem for terrestrial vegetation, playing a key role in the global reduction–oxidation cycles of carbon and nitrogen (Chapin et al., 2011). Soil systems are subject to natural changes, including both directional and cyclic changes that occur over time scales ranging from days to millennia.

Soil properties result from the dynamic balance of soil formation (it can take up to 1000 years to form 1 cm of soil, Wall & Six, 2015) and soil loss. Soil formation depends on the balance between soil development, deposition, and erosion (Chapin et al., 2011), and was originally governed by at least five independent control variables: climate, topography, parent material, potential biota and time (Amundson & Jenny, 1997). For thousands of years, humans have altered soils, but this influence has greatly increased since the early twentieth century. Humans are now an important agent of soil formation (Schmidt et al., 2014) and alteration, and soils around the world have been irrevocably altered (Amundson & Jenny, 1997) – a process called soil degradation. Irreversible loss of soil is a result of human depletion, including soil erosion, salinization, and other degradation processes. Human-induced soil degradation in America with high and very high severity occurs mainly in Mesoamerica and Caribbean, but extensive land areas are also on human-induced soil degradation in both North America and South America, notably due to agriculture (Karlen & Rice, 2015, see Chapter 4).

Soil is the largest terrestrial carbon pool (Scharlemann et al., 2014). Carbon content in the topsoil in the Americas ranges from 2 to 3% by weight (Figure 2.21), although some soil types (e.g. wetlands, peatlands) have much higher soil carbon content. Soil degradation and changes in land use can strongly affect its capacity to store carbon (section 2.2.9).

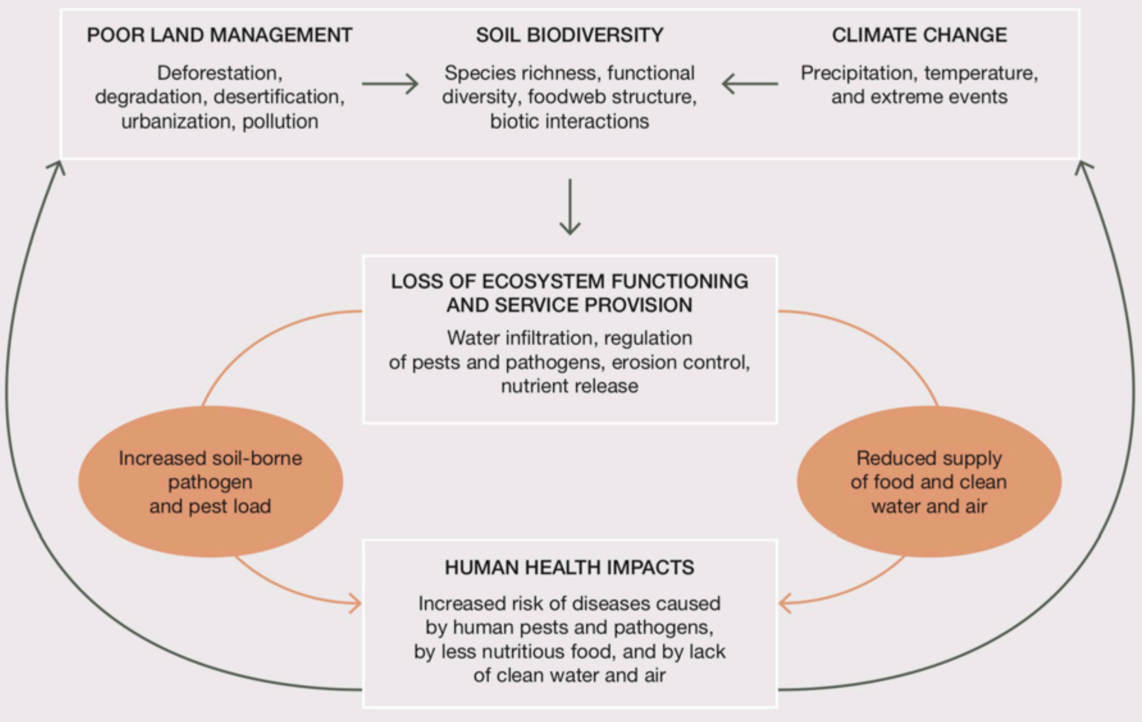
Figure 2 21 Average carbon content in topsoil expressed as percent by weight in subregions of the America. Source: FAO (2018). FAOSTAT. Average carbon content in the topsoil as a % in weight. http://data.fao.org/ref/fd1ee060-9eb8-4b39-bf25-0bca-c38ad597.html?version=1.0 date Accessed: May 22, 2018.



According to Guo & Gifford (2002), land use change can either reduce or increase carbon storage up to 80%. In general, changes in forest to crop lands can reduce carbon storage by 40% and a change from pasture to crops can reduce it by up to 60%. On the other hand, soil carbon stocks can increase after some types of land use change, for example from native forest to pasture (~10%), crop to pasture (~20%) and crop to plantation (~60%). The increase in soil organic matter stocks may be due to several factors, including (i) the large amount of fine roots which contribute to the reduction of water and gas exchange, decreasing soil organic carbon decomposition rates and (ii) the fact that soil under pasture is not disturbed (plowed) as are croplands among others (Rittl et al., 2017)

Soil biodiversity is another component strongly affected by human-induced changes (Wall et al., 2015). These changes may have a cascade effect on soil diversity and extend beyond the site of the disturbance (Haddad et al., 2015). Wall et al. (2015) suggest that reducing soil biodiversity may lead to an increased risk of diseases caused by: (i) human pests and pathogens, (ii) less nutritious foods, and (iii) lack of water for the environment (Figure 2.22) as shown below. Soil biodiversity can be maintained and partially restored if well managed. Good soil biodiversity management practices should focus on maintenance of amount and quality of soil organic matter and the prevention of soil erosion. Additionally, agricultural systems with fewer inputs may promote self-regulating systems and higher biodiversity (Thiele-Bruhn et al., 2012). This is particularly important to approach the SDG2 (Zero hunger), SDG3 (Good health and well-being), SDG12 (Responsible production and consumption) and SDG15 (Life on land). Maintaining the soil-clean water-clean air dynamic is fundamental to food production, the quality and quantity of water and its effects on food security and quality of life (Wall & Six, 2015; Wall et al., 2015).

Figure 2 22 **Links between soil biodiversity and human health. Source: Wall *et al.* (2015).**



2.3 Effects of trends in nature's contributions to people on quality of life

Links between NCP and quality of life have been conceptually described in many instances (e.g. Diaz et al., 2015; Pascual et al., 2017). Nevertheless, to our knowledge, a clear picture of what bundles of NCP contribute to each aspect of well-being has not been shown. In this sense, our team performed a Delphi process (Hasson et al., 2000; Landeta, 2006), which relied on a panel of experts (11 leading authors of this chapters) to build consensus through interactive rounds of scoring the links between each of the 18 NCP and six elements of quality of life: food security, water security, energy security, health, livelihood security (as well as securing ways of living), and experiencing nature (e.g. the emotional and spiritual securities that may contribute, for instance, to cultural continuity). The emerging picture is presented in Table 2.15. In the following subsections, each of these six elements of quality of life will be discussed in turn with the focus on the SDGs, supporting the 2030 Agenda of the United Nations Development Program, as well as the CBD Aichi targets.

Table 2.15. The relationship between 18 of nature's contributiosn to people (NCP) and six elements of a good quality of life were assessed based on the expert opinion of 11 chapter lead authors, using the Delphi method. Mean valuesstandard error of a two-round scoring exercise are reported. Authors were asked to evaluate the strength of the relationships with 0 = none, 1 = weak, 2 = moderate and 3 = high.

Table 2 15 **The relationships between 18 of Nature's Contributions to People (NCP) and 6 elements of a good quality of life.**

Mean values [standard error] were assessed based on the expert opinion of 11 chapter lead authors, using the Delphi method with a two-round scoring exercise. Authors were asked to evaluate the relationships between NCP and aspects of a good quality of life as: 0 = none, 1 = weak, 2 = moderate and 3 = high.

Nature's Contribution to People	Food Security Mean	Water Security Mean	Energy Security Mean	Health Mean	Livelihood Security Mean	Experiencing Nature Mean
Food and feed	3.0[a]	2.1[c]	1.6[c]	2.9[b]	3.0[a]	1.6[c]
Materials and assistance	1.5[c]	1.5[c]	1.5[c]	2.2[c]	2.6[c]	2.0[b]
Energy	1.6[c]	2.0[d]	3.0[a]	1.9[c]	2.5[c]	0.9[a]
Medicinal, biochemical and genetic resources	2.1[d]	1.0[c]	0.9[b]	2.8[b]	2.4[c]	1.5[c]
Learning and experiences	1.5[d]	1.3[d]	1.0[d]	2.3[c]	2.6[c]	2.9[b]
Supporting identities	1.9[d]	1.5[e]	0.9[c]	1.8[d]	2.7[c]	2.4[c]
Physical and psychological experiences	1.1[d]	1.2[d]	0.6[c]	2.3[d]	2.4[c]	2.7[c]
Maintenance of options	2.2[c]	1.6[d]	1.7[c]	2.6[c]	2.1[c]	2.1[b]
Regulation of climate	2.1[d]	2.5[c]	1.7[c]	2.1[d]	1.6[c]	1.3[c]
Regulation of freshwater quantity, flow and timing	2.7[b]	3.0[a]	1.5[d]	2.7[b]	2.1[b]	1.6[c]
Regulation of freshwater and coastal water quality	2.6[c]	2.8[b]	1.0[d]	2.5[c]	2.1[b]	1.5[c]
Regulation of hazards and extreme events	1.8[c]	2.2[b]	1.4[c]	2.5[c]	2.4[c]	1.3[c]
Habitat creation and maintenance	2.1[b]	1.8[b]	0.9[c]	1.5[d]	1.5[c]	2.2[c]
Regulation of air quality	0.9[c]	0.7[b]	1.2[c]	3.0[a]	1.8[c]	1.8[c]
Regulation of organisms detrimental to humans	2.4[c]	1.7[d]	0.8[c]	2.8[b]	1.7[c]	1.2[c]
Pollination and dispersal of seeds and other propagules	2.9[b]	0.8[c]	0.5[c]	1.9[c]	1.6[c]	1.6[c]
Regulation of ocean acidification	2.1[c]	1.5[d]	1.0[e]	1.2[d]	1.6[d]	1.0[d]
Formation, protection & decontamination of soils/sediments	2.3[c]	1.6[d]	1.0[d]	1.3[d]	1.7[d]	1.0[d]

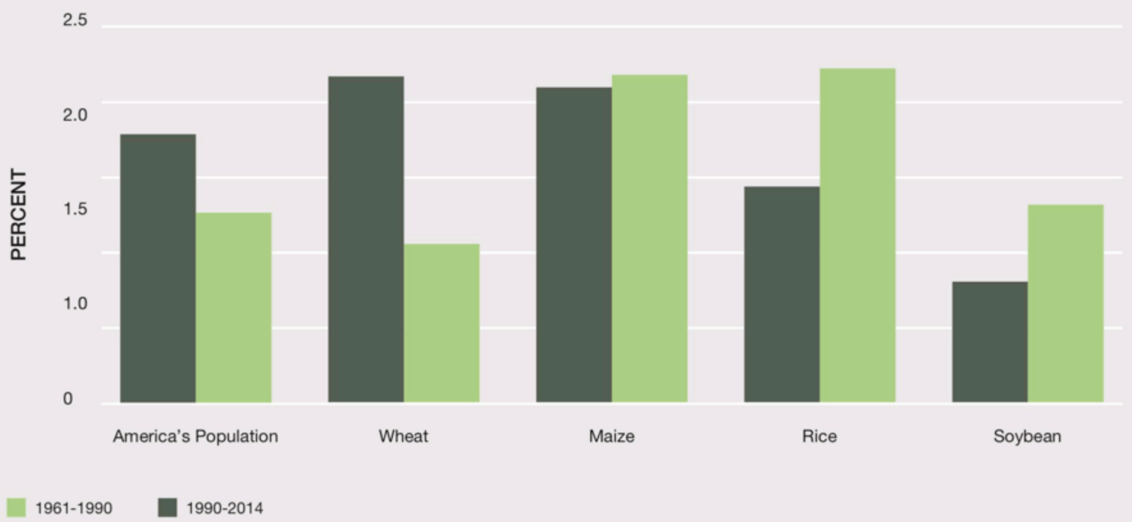
VALUE	STRENGTH	SE	LEGEND
0	None	0.0	a
0.1 — 1	Weak	0.1	b
1.1 — 2	Moderate	0.2	c
2.1 — 3	High	0.3	d
		0.4	e

2.3.1 Food Security

Food is an essential part of human well-being. It provides us the energy and nutrients, and ensuring access to food is crucial to achieve a healthy and productive life. Food security is "a situation that exists when all people, at all times, have physical, social and economic access to sufficient, safe and nutritious food that meets their dietary needs and food preferences for an active and healthy life" (FAO, 2006) and this concept is determined by three factors: availability, access to adequate resources or entitlements, and utilization of food through an adequate diet. Those factors are intrinsically hierarchical, in which availability is required but not sufficient to ensure access, which is not necessarily stable, and may not be sufficient for effective utilization (Barrett, 2010). However, this concept has been in part criticized for being too narrow, and failing to consider other relevant factors, including policy, equity, and diversity (Wittman et al., 2016), as well as for not taking full account of the traditional food practices (Power, 2008). Recently, the ethical and human right dimension of food has come into focus through the SDG 2, which aims to end all forms of hunger and malnutrition by 2030, making sure all people – especially children – have access to sufficient and nutritious food all year round (UN, 2016). Food production systems are supported by the services provided by natural ecosystems, such as pollination, biological pest control, maintenance of soil structure and fertility, nutrient cycling and hydrological services. In turn, this activity also generates ecosystem services such as soil regulation, climate stabilization through greenhouse gas mitigation, biodiversity support and water purification, but it is also responsible for damage such as, nutrient pollution, biodiversity loss, and greenhouse gas emissions (Power, 2010; Pretty, 2008; Robertson et al., 2014; Stallman, 2011). Historically the availability of food has increased thanks to advances in agricultural production, and increased yields. According Schmitz et al. (2014) during the pre-industrial period, cropland expansion was the source of increased agricultural production, but since the mid-20th century intensification through new technologies is the main cause of growth. As a result, while the area of cropland increased by about 15% worldwide between 1955 and 2005, agricultural production increased by more than 200%.

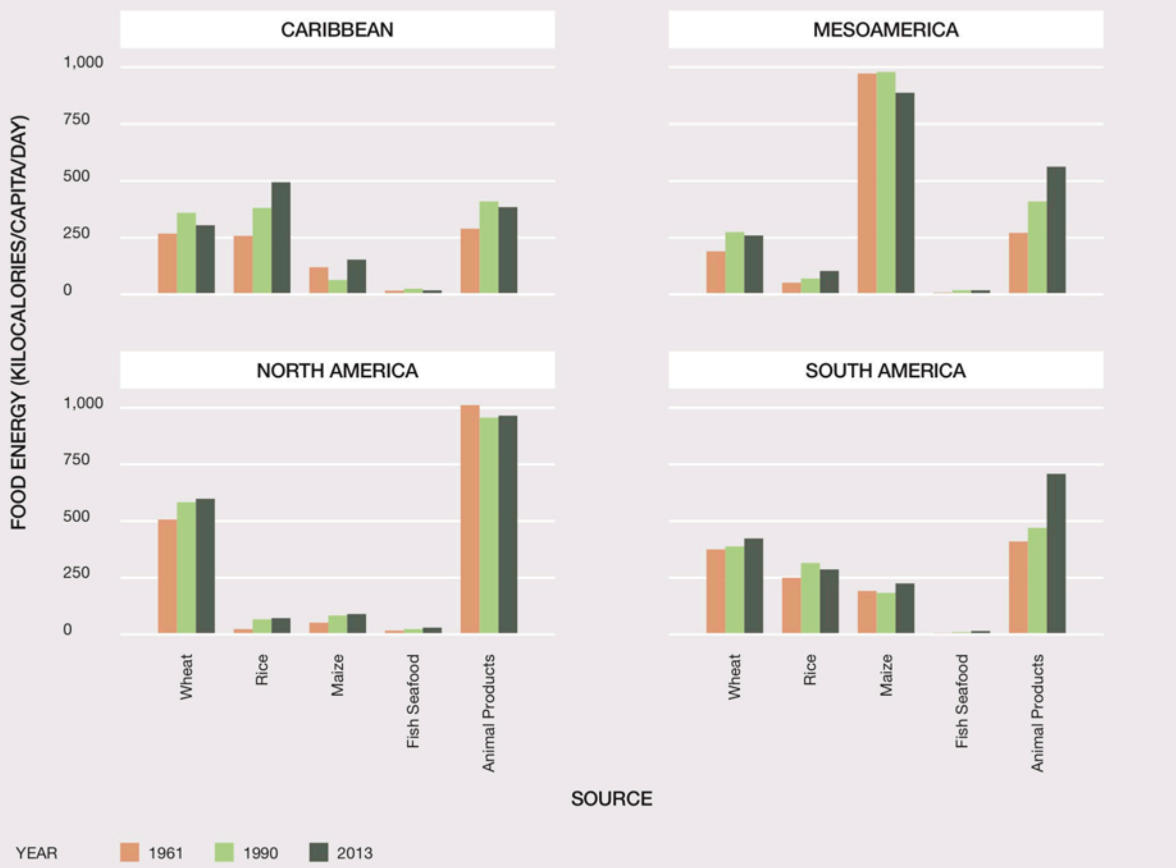
Although some crop yields are increasing faster than the rate of population growth (see Figure 2.23) much of the agricultural production is exported to other regions or devoted to other sectors. For example, soybeans have become one of the most important agricultural commodity in Brazil (Pashaei Kamali et al., 2016) and Argentina (Vazquez et al., 2017), due to an increase in global demand for soybean flour and oil of which about 70% of the production is exported to China (USDA, 2016). In the last 10 years, the use of maize for fuel production has increased, accounting for approximately 40% of the maize production in USA, and affecting maize prices for animal and human consumption (Ranum et al., 2014). Wheat, maize, rice and soybean are projected to provide 85% of the increase in food cereal consumption to 2050, and maize and soybean will continue to provide the animal food calories, converting crops into secondary protein supply for humans (Fischer et al., 2014). However, increasing yields will address only one aspect (availability) of food security, which requires multiple approaches and solutions (Poppy et al., 2014).

Figure 2 23 **Average annual growth rate (%) for the Americas population and crop yield increases. Source: FAO (2017). FAOSTAT Statistics Database. http://www.fao.org/faostat/en/#data/QC. Date accessed: January 20, 2017.**



In the Americas region, per capita calorie and protein demand increased during the period between 1961 and 2013, mainly from animal products in developing countries (Figure 2.24) showing differences between regions. Products from livestock are the principal protein source in all regions, and according to Sans & Combris (2015) meat consumption has surged over the last 50 years, rising from 61 g per person per day in 1961 to 80 g per person per day in 2011 worldwide. In Mesoamerica, maize has been the major source of food energy, where the highest consumption was 267 and 187 gram per capita per day in Mexico and Guatemala, respectively. During the last 30 years in the USA, 43% of maize production is used to feed animals to support the high consumption of animal products (Ranum et al., 2014). In North America, wheat has historically been the main source of calories and protein from cereals, whereas rice has the highest importance in the Caribbean region, and both wheat and rice rank high as cereal food sources in South America. Despite the low per capita intake of fish and shellfish, when data are aggregated by subregion, fish and shellfish are vital for food security in coastal area and in wetlands of all subregions of the Americas, particularly for small-scale fishers and their families (Béné et al., 2007), as fish is considered the healthiest sources of animal protein for human consumption (section 2.2.1.3 for more details).

Figure 2 24 Components of average food supply in 1961, 1990 and 2013, for the America's region. Source: FAO (2017). FAOSTAT Statistics Database. http://www.fao.org/faostat/en/#data/FBS. Date accessed: April 15, 2017.



The Americas' population is projected to reach 1.2 billion by 2050, an increase of 22.7% over the current population (UN Department of Economic and Social Affairs, Population Division, 2015). Although as a region, the Americas has largely overcome food insecurity during the last few decades, differences exist between countries and subregions. Undernourishment in Mesoamerica, the Caribbean and South America has been reduced from 14.7% to 5.5% in the past 20 years, but over 40.7 million people remain undernourished in Latin America and 3.6 million face severe food insecurity in North America (FAO, 2017, p. 90). In North America, food insecurity is linked to households with incomes near or below the federal poverty line, households with children headed by single women or single men, women and men living alone, and black- and hispanic-headed households (Coleman-Jensen et al., 2016). In Mesoamerica, the percentage of undernourished people has declined from 14.7% in 1990–92 to 5.5% in 2014–16 (FAO, IFAD & WFP, 2015), but in the Caribbean this proportion still varies widely, with the average at a staggering 19.8% (Table 2.16). Progress in reducing poor nutrition is associated with good economic performance, growth in the agricultural sector, and social protection policies, and not because of better household income alone.

According to Barret (2010), most severe food insecurity is associated with chronic poverty. In the Americas, agriculture is the principal driving force of the rural economy and, for those developing countries without substantial mineral resources, often the whole economy. Agriculture on its own can lead to growth in countries with a high share of agriculture in GDP (FAO, 2015). Nevertheless, in the poorest areas of Latin America (northeastern Brazil, southern Mexico, the Andes, and the densely settled hillside areas of Central America and the Caribbean), rural poverty, population growth, and unsustainable agricultural systems are leading to the degradation of many NCP as well as the breakdown of indigenous communities and their natural resource management systems (Pichon & Uquillas, 1997). Then, more sustainable forms of land use and efficient agricultural production systems (Pretty, 2010) are needed for poverty reduction and improve the food security.

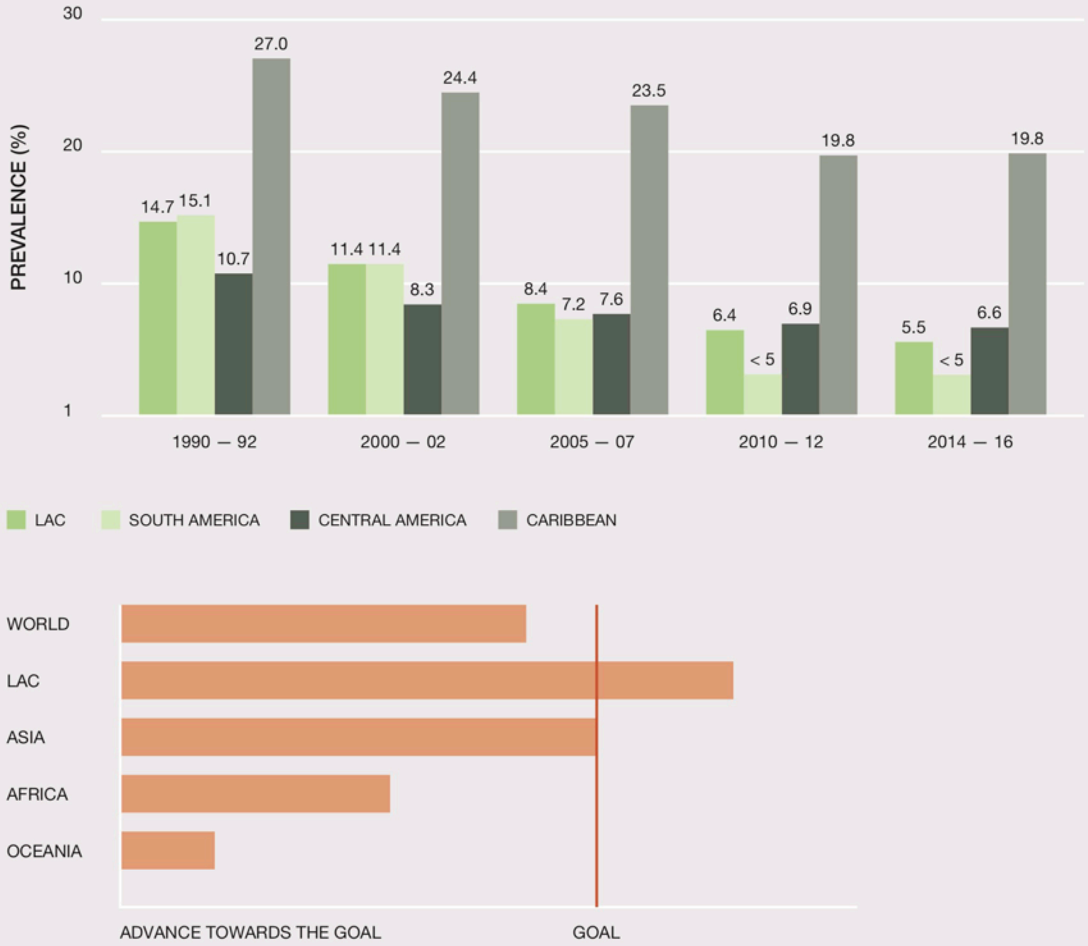
Table 2.16. Prevalence of undernourishment in the Americas.

Subregions	**Number of people undernourished**								**Proportion of undernourished in total population (%)**						
	1990-92	**2000-02**	**2005-07**	**2010-12**	**2014-16**		**Change 1990-2016**		**1990-92**	**2000-02**	**2005-07**	**2010-12**	**2014-16**		**Change 1990-2016**
	(millions)						**%**		**%**						**%**
North America									<5.0	<5.0	<5.0	<5.0	<5.0		na
Mesoamerica	12.6	11.8	11.6	11.3	11.4		–9.6		10.7	8.3	7.6	6.9	6.6		**–38.2**
Caribbean	8.1	8.2	8.3	7.3	7.5		–7.2		27	24.4	23.5	19.8	19.8		**–26.6**
South America	45.4	40.3	27.2	ns	ns		<–50.0		15.1	11.4	7.2	<5.0	<5.0		**na**
na: not applicable; ns: not statistically significant. Source: FAO *et al.* (2015)															

The Global Food Security Index (http://foodsecurityindex.eiu.com/) addresses the issues of affordability, availability and utilization to understand the risk of food security in countries and regions. The countries are grouped in quartiles based on: best environment, good environment, moderate environment and those that need improvement. Based on the analysis of this index, the North American subregion and Canada and the USA at the country-level do not face food insecurity, but certainly certain social sectors still face hunger and malnutrition (Table 2.16). Data from the FAO (Figure 2.25) indicate that Mesoamerica, the Caribbean and South America are making greater progress towards food security and achieving SDG 2 (Zero hunger), compared to other global regions. Yet, the Caribbean subregion is still particularly vulnerable and some countries in Mesoamerica and South America maintain high levels of food insecurity.

Considering the role of family farmers in food security, Graeub et al. (2016) provide a rough estimate of the calorie requirements in each country currently being supplied by family farmers. While countries in North America and Mesoamerica are, on average, 60% sufficient, South America achieved only 36% sufficiency (the lowest level of any world region).

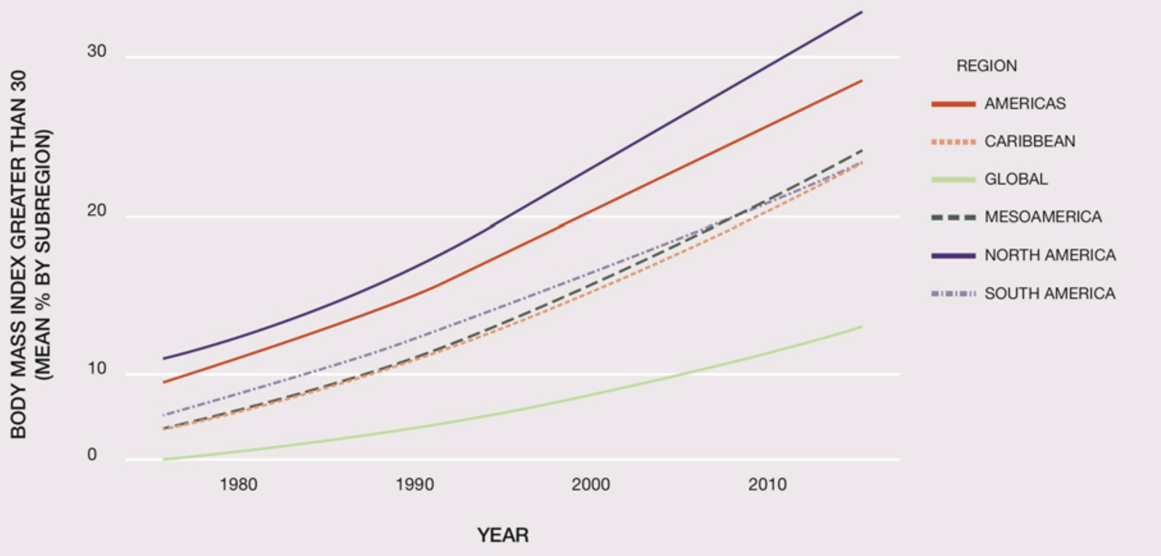
Figure 2 25 A Prevalence of hunger in Latin America and the Caribbean (LAC);
B State of progress towards the 1C goal of Millennium Development Goals in the subregions of Latin America and the Caribbeanthat (Millennium Development Goal 1C: to halve the proportion of individuals suffering from hunger in the period between 1990 and 2015). Source FAO (2015).



Food consumption patterns also showed a transition. During the last decades, they passed from the high prevalence of under-nutrition to over-nutrition (Kearney, 2010). Drivers such as the increase in the human population, rural-to-urban migration and income increases in developing countries created challenges in managing a balanced diet (Nantapo et al., 2015); in addition, the increased intake of high sugar and high fat foods characteristic of modern diets lead to a growing number of diseases associated with unbalanced nutrition, such as obesity (Figure 2.26) and diabetes (Pretty et al., 2010). Hence, this leads to the greater pressure on the food supply system, increased competition among food producers for land, water and energy, ultimately leading to negative effects on the environment (Godfray et al., 2010). According to World Health Organization, the cause of obesity and being overweight is an increased intake of energy-dense foods high in fat and an increase in physical inactivity associated to urban lifestyles and modes of transportation. Today, obesity has reached epidemic magnitudes worldwide, with at least 2.8 million people dying each year because of being overweight or obese (WHO, 2017). Findings suggest that the increase in food energy supply explains the increase in average population body weight, mainly for high-income countries (Vandevijvere et al., 2015), but now also for low and middle-income countries.

Figure 2 26 **Trends in prevalence of obesity among adults for the Americas 1975–2016.**

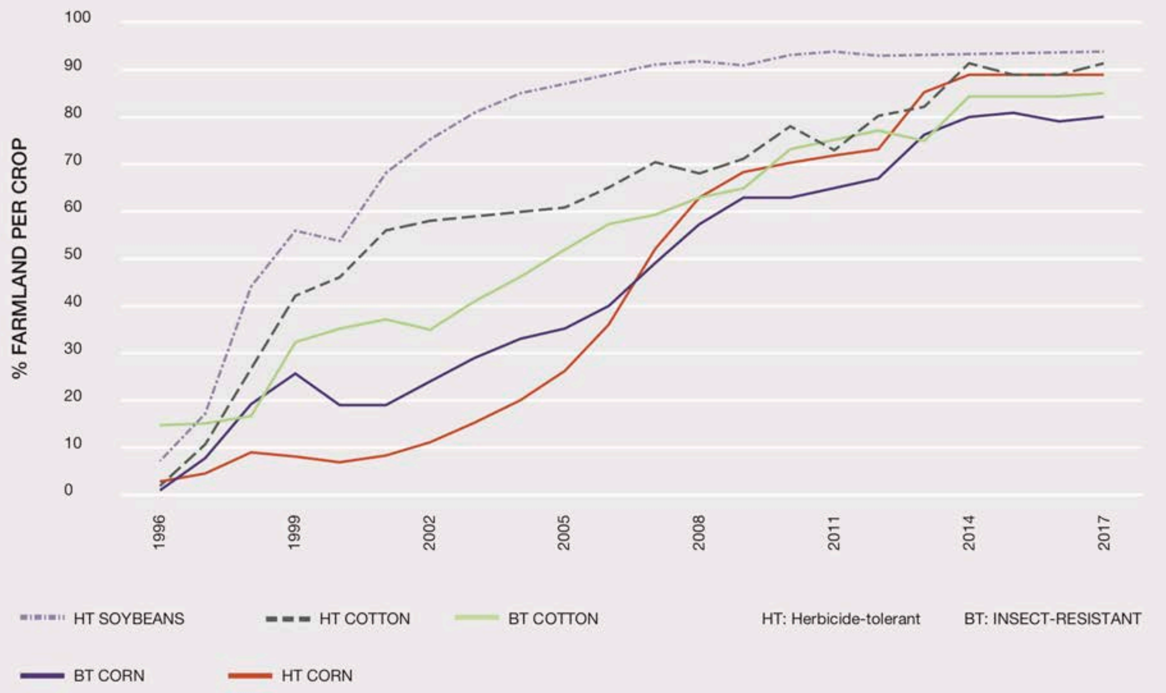
Body Mass Index ≥ 30, mean based on % of estimates by country. Source: WHO Global Health Observatory data repository (2017). http://apps.who.int/gho/data/view.main.CTRY2450A?lang=en.



Another important issue of concern regarding food security regards the production of Genetically Modified Organisms (GMO). In the Americas, GMO technology is widespread in USA, Brazil and Canada. For instance, the production of GMO soybean, cotton and corn skyrocketed in the past two decades (Figure 2.27). Nevertheless, there is concern about the safety of GMOs, and there is no scientific consensus on GMO safety to date (Hilbeck et al., 2015).

Figure 2 27 **Adoption of GMO crops in USA as percentage (%) of farmland.**

Source: USDA, Economic Research Service using data from Fernandez-Cornejo and McBride (2002) for the years 1996–1999 and USDA, National Agricultural Statistics Service, June Agricultural Survey for the years 2000–2017. https://www.ers.usda.gov/data-products/adoption-of-genetically-engineered-crops-in-the-us/recent-trends-in-ge-adoption.aspx. Accessed: December 2017.



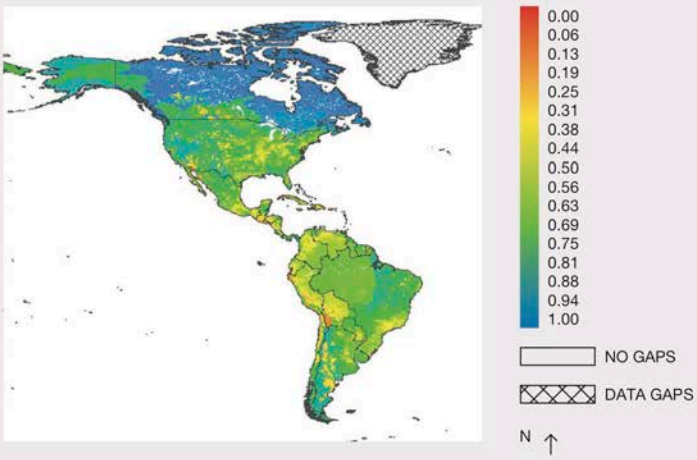
2.3.2 Water security

Water is vital for the survival of humans and ecosystems. It is fundamental for material, recreational, aesthetic, and spiritual basis of life, and ensuring access to safe water is essential for the provision of other rights such as food, health and welfare. Water security refers to a supply of water of adequate quantity and quality needed to sustain health, livelihoods, economic development, and ecosystems, and protect against water-borne pollution and water-related disasters (UN, 2013; Grey & Sadoff, 2007). The finite amount of available freshwater can limit the progress made towards the three dimensions of sustainable development (social, economic, environmental), and this constraint is increasing. Sustainable Development Goal 6 addresses this, with the target of ensuring the availability and sustainable management of clean water and sanitation by 2030, taking into account issues related to both water quantity and quality (UN, 2016). SDG 6 is, in turn, linked or relevant to the Aichi Biodiversity targets 8, 11, 14, 15, which refer to water security as an essential element of quality of life.

Water insecurity occurs when human and environmental factors create variability in the availability of water, relative to its need. Factors such as climate change, population growth, changing land use (increasing agricultural and urban lands), and pollutants alter water quality and affect its supply; this is exacerbated by economic disparity and poor governance and in some cases, for example in Nunavut, Canada, the loss of indigenous knowledge systems (Gain et al., 2016; Vorosmarty et al., 2015; Medeiros et al., 2017). Water security index scores, which are based on indicators derived from SDG 6, indicate relatively high scores for most of the Americas, with low scores in western Peru and southern Bolivia (Figure 2.28). In some areas facing water scarcity (e.g. portions the southwestern USA and Mexico) water security index scores are higher than predicted due to the mitigating effects of technology based water management. Engineering solutions to replace the ecosystem services can be effective but expensive, and tend to rely on fossil fuels. A heavy dependence on technology to meet water demands may produce false security. Desalination plants are one example. In northwest Mexico desalination plants are in widespread use despite their high environmental impacts, including those to adjacent marine ecosystems where plant effluents decrease coastal productivity and the livelihoods of local communities (Cortes et al., 2012; McEvoy & Wilder, 2012).

Figure 2 28 **An aggregate water security index based on measuresof water availability, accessibility, safety and quantity and management throughout the Americas**

Scores range from 0–1, representing a continuum of low to high security. Grey shaded areas are data gaps. Source: Gain *et al.* (2016).

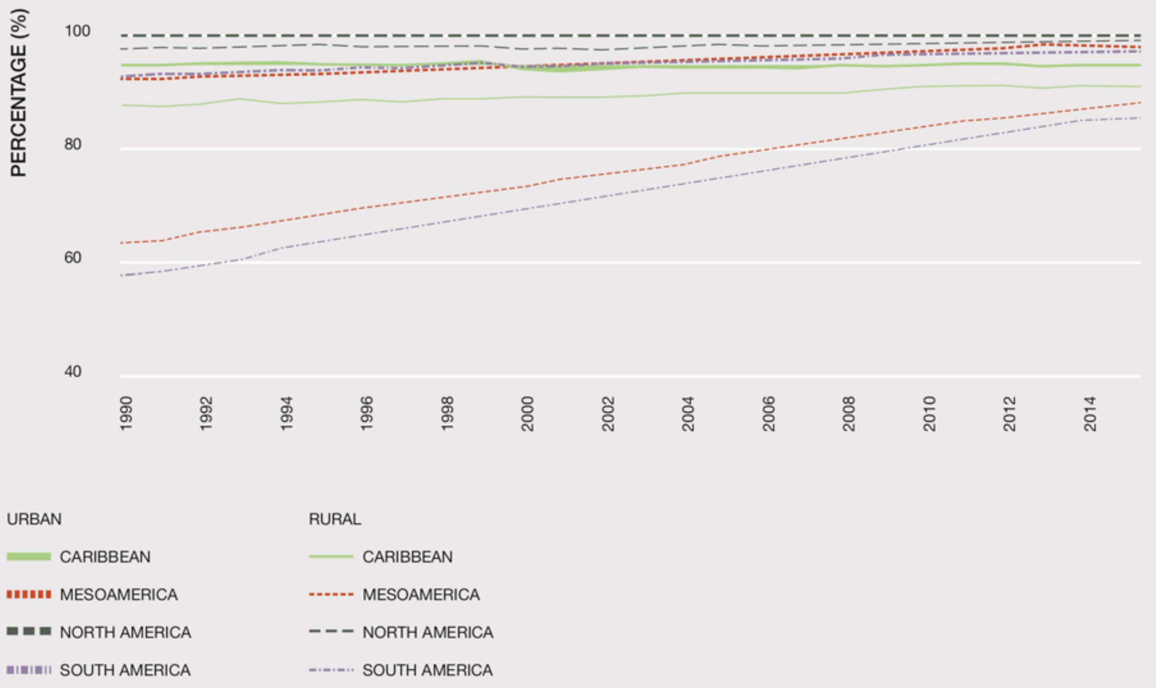


Considerable progress has been made in the Americas in providing access to improved water sources since 1990, with access in nearly all regions at or above 90% of the population; only the rural populations in South and Mesoamerica lag at about 80% (Figure 2.29). The proportion of the rural poor in South America with access to potable water has increased only 10% in 30 years, leaving 18% without access in 2015. Overall demands for water in the Americas are increasing, particularly in areas of high economic development. At the same time, human activities lead to increasing pollutant loads that compromise both the support of biodiversity and the safety of water supplies. Assessments of water demands by subregion mask the high degree of spatial variability in water demand for production, and the threats they pose to water security (Hoekstra et al., 2012).

Access to sanitation and hygiene is also a key part of the definition of water security. As a right of all humans it helps safeguard health and well-being, and is a key in alleviating poverty. Access to sanitation is increasing, although it lags behind access to improved drinking water sources, particularly for rural populations and the poor. There are still 2.4 billion people globally who lack access to basic sanitation services, such as toilets or latrine (WWAP, 2015).

Figure 2 29 **The proportion of rural and urban populations in the Americas by subregion with access to improved water sources between 1990 and 2015.**

Source: Own representation of data from World Bank (2017). World Development Indicators. https://data.worldbank.org/data-catalog/world-development-indicators. Improved water source, urban and rural (% of urban population with access, % of rural population with access). Last updated January 3, 2017.



Water use in the Americas is dominated by agricultural needs (Table 2.17, Gleick, 2014). In Mesoamerica, South America and the Caribbean, about 74% of freshwater is used for agriculture and domestic use is the second largest consumer. In contrast, use in North America is dominated by industry, largely due to water withdrawals for industrial cooling, for example by power plants, that account for almost 50% of freshwater withdrawals. This use is not consumptive; consumptive uses of water by industry are much lower (Kenny et al., 2009; Gleick, 2014). Forecasting future water withdrawals under climate change indicates increasing future demand in South America, mostly for industrial use, and relatively stable future demands in North America (IPCC, 2002; Alcamo et al., 2007). Locally non-sustainable use of freshwater, such as the withdrawal of fossil groundwater from aquifers with no long-term net recharge for irrigation is common in the arid regions, particularly in North America (Shilkomanov & Rodda, 2003). The use of groundwater in the USA has greatly increased food production and been a source of water for decades, providing drinking water for about half the total population and nearly all of the rural population, as well as providing over 50 billion gallons per day for agricultural needs. However, it’s cumulative depletion, for example between 1900 and 2008 was about 1,000 km^3—equivalent to about twice the water volume of Lake Erie (Konikow, 2013) - now poses a threat to water security as aquifers are drawn down, particularly in the plain states (Vorosmarty et al., 2010).

Nature's provision of freshwater supplies often requires human intervention to withdraw, divert, and transport water to engineered storage sites and sites of use (Viessman & Hammer, 2008; Verma et al., 2015). Many wells, human-made water supply impoundments, and water distribution systems have been constructed to increase the reliability of freshwater supply (Cech, 2010), although these may reduce water available to support biodiversity. Dam building and the creation of other structural facilities (e.g. canals, channels and pipes) are a common means to manage water supplies and stabilize flows (Grigg, 2005), although dams are increasingly recognized for their heavy impacts on other NCP (Palmer, 2010). The impacts to local communities are equally as great through the disruption or displacement of local communities, a loss of a sense of belonging, and loss of farmland and cultural heritage (Tucker et al., 2016)

Table 2.17. Water withdrawals (cubic kilometers/year and %) by use category. The highest use in each subregion is highlighted.

Region	Municipal Use		Industrial Use		Agricultural Use		Total
	$km^3 yr^{-1}$	% of total	$km^3 yr^{-1}$	% of total	$km^3 yr^{-1}$	% of total	$km^3 yr^{-1}$
North America	68.0	13.0	281.5	53.1	179.8	34.6	610.0
Mesoamerica	14.7	16.0	8.6	9.0	79.1	75.0	92.4
Caribbean	4.4	21.0	4.7	22.4	12.1	67.6	21.1
South America	36.0	17.0	26.0	12.0	154	71.0	216.0
Source: FAO Aquastat (2015)							

Human appropriation of freshwater supplies (water volume consumed) can be assessed using the water footprint (Figure 2.30), represented by three components: blue water (the surface and ground water consumed, for example in irrigated agriculture, industry, and domestic), green water (rainwater stored in soil that is consumed, e.g. in crop production) and grey water (freshwater required to assimilate waste using existing water quality standards; Figure 2.31). The global water footprint between 1996-2005 was 9,087 billion cubic meters per year; agricultural production contributes 92% to the total footprint.

A substantial portion (20%) of the global water footprint supports agricultural production for export to other countries, or *virtual water* (the flow of water hidden in food and other commodities). This allows water poor regions to support larger human populations by importing water intensive crops, thereby preserving local water resources. Subregions in the Americas tend to be major water exporters, particularly the USA, Brazil, Argentina, and Canada (Figure 2.32, Mekonnen & Hoekstra, 2011).

Figure 2 30 **Trends in the total water footprint by subregion in the Americas. Source: Water Footprint Network.**

Visuals prepared by the IPBES Task Group on Indicators and Knowledge and Data Technical Support Unit based on raw data provided by indicator holders. Only for IPBES assessment and TGI – approved use – please do not distribute.

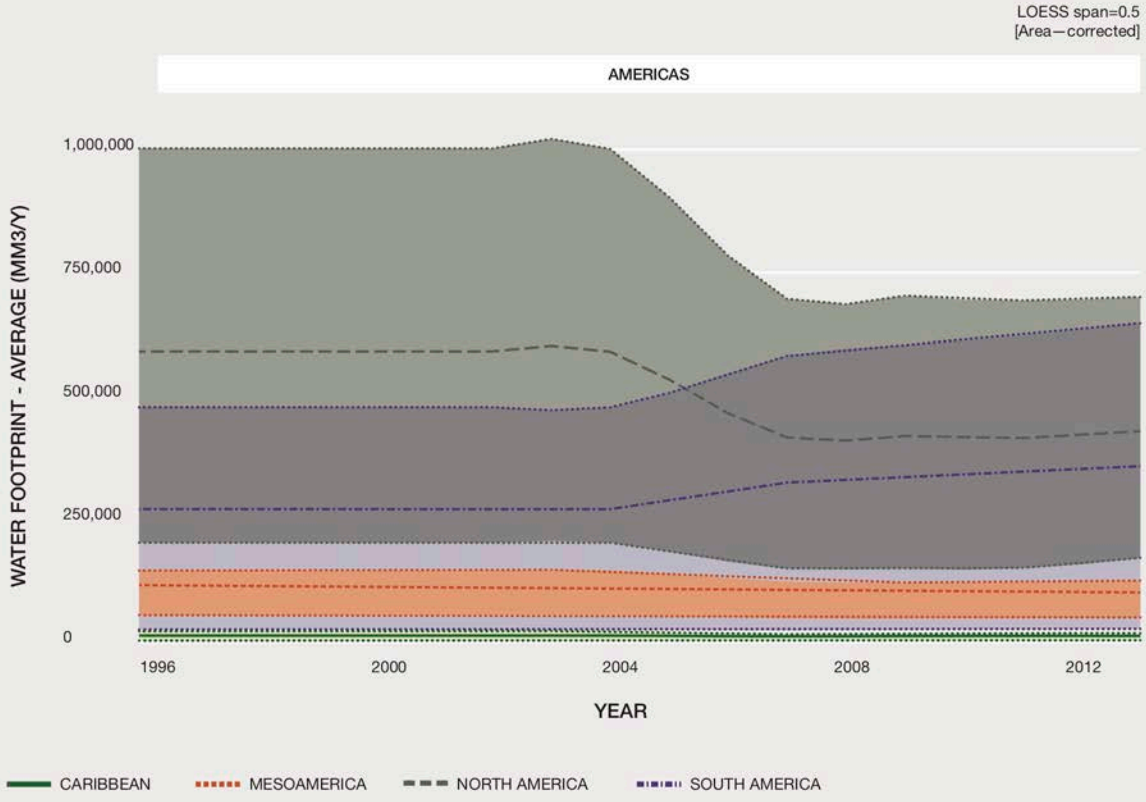


Figure 2 31 **Water footprint for the Americas by three components, represented as green, blue and grey.**

The blue water footprint includes consumption of surface and ground water (i.e. blue water resources), green is the volume of rainwater consumed (e.g., in crop production), grey encompasses the volume of freshwater required to assimilate pollutants based on existing water quality standards. Black is the total. Source: Mekonnen & Hoekstra (2011).

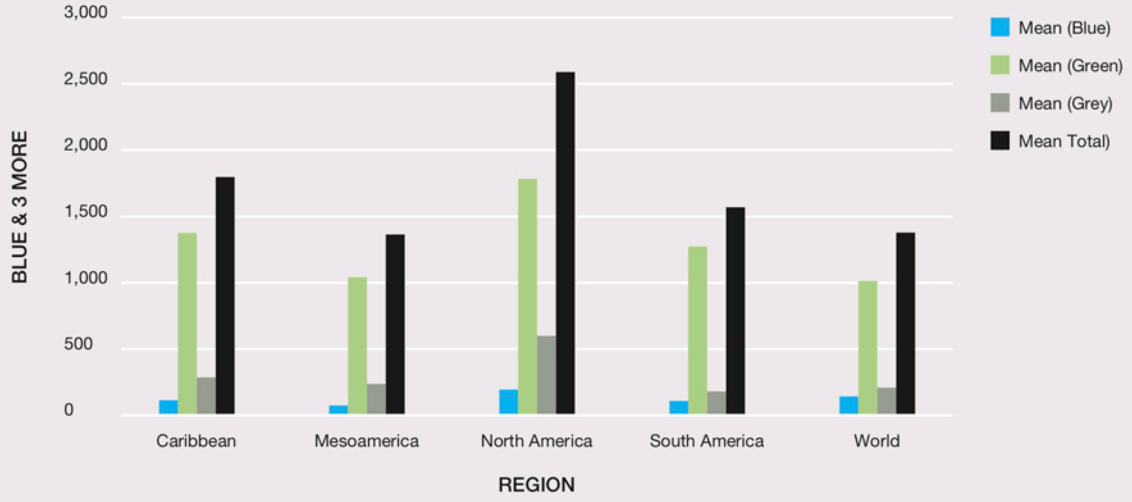
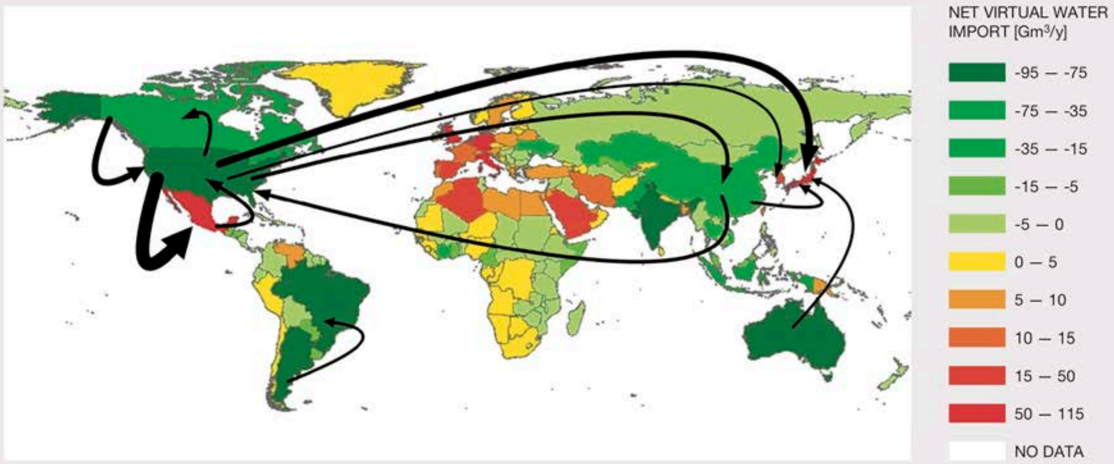


Figure 2 32 **Net virtual water export from countries of the Americas through agricultural and industrial goods between 1996–2005.**

Countries shown in green are net virtual water exporters; those in yellow and red import virtual water. The biggest net exporters are the USA, Canada Brazil, and Argentina (calculations made using only gross flows over 15 billion m^3 per year; the size of the arrow indicates relative flow), and the largest importer is Mexico. Source: Hoekstra & Mekonnen (2012).



Effective water management is dependent on effective land management (Bossio et al., 2010). Natural ecosystems such as forests, wetlands, riparian zones and floodplains, and grasslands help to maintain water quality through filtration, groundwater renewal, and maintenance of natural flows (Honda & Durigan, 2016). The loss of natural ecosystems reduces their benefits, presenting risks to human health in the form of decreased drinking water quality, higher water costs that have a greater impact on the poor, and decreases in crop yields and hydroelectric power due to reduced flows in the dry season (Postel & Thompson, 2005). The benefits of maintaining healthy watersheds to preserve water supplies are well supported, although insufficient, across the Americas. For example, in Quito, Ecuador, more than 1.5 million residents receive drinking water from two protected areas in the Andes, the Cayambe - Coca (400,00 ha) and Antisana ecological reserves (120,000 ha). As part of Ecuador's National Parks System, these areas are also used for cattle, dairy, and timber production, and supports a human population of 27,000. A trust fund was established to provide payments to landowners in return for their work to protect water quality (Pagiola et al., 2002). In the Dominican Republic, the Madre de las Aguas Conservation Area conserves the source of 17 rivers that provide water for irrigation and domestic use to over half of the country's population. The area makes up about 5% of the area of the Dominican Republic, and contains many local stakeholders that live in small rural communities. It also supports a rich diversity of species. The protected watershed of Banff National Park in Canada flows into Alberta's Bow River Basin that is home to 1.2 million people. These areas support mixed use while supplying drinking water, for example by providing recreational opportunities and support to farmers and industries (WCPA, 2012). In Central America, the city of Tegucigalpa in Honduras is one of several large Latin American cities that protect surrounding cloud forest to guarantee water supplies, in this case in the La Tigra National Park (Hamilton, 2008). In Costa Rica, *Agua Tica* is an initiative to contribute to water security of the Greater Metropolitan Area of San Jose through watershed conservation. In California, USA, around 85% of San Francisco's drinking water comes from snowmelt captured in a reservoir in Yosemite National Park (CBD, 2008). In nearly all of these cases (and there are other examples) the beneficiaries of the water supply services are not the same as those who bear the costs of providing those benefits. Linking the providers to the beneficiaries is important in designing the mechanisms that make agreements possible, for example direct payments and/or trust funds to provide grants to support environmentally sustainable development projects for communities in the watersheds. Thus, watershed protection can be integrated with rural development and livelihoods (Postel & Thompson, 2005).

Wetlands are particularly effective at regulating flows and purifying water. The Ramsar Convention lists wetlands that have been designated as Wetlands of International Importance and promotes their wise use in the context of sustainable development to benefit people and biodiversity (www.ramsar.org). Nearly 650,000 km^2 of wetlands have been designated as internationally important in the Americas, over half of which is in South America (see Figure 2.16). This reflects their links to food security and livelihoods, for example over 660 million people globally rely on fishing and aquaculture for a living, and many fish species reproduce in coastal wetlands, contributing to dietary diversity. Ramsar wetlands also allow for continued socio-cultural traditions and income generation (Horwitz et al., 2012). For this reason, Mexico proposed and the Ramsar Convention accepted in its 12th Conference of the Parties (Uruguay 2015) the Resolution XII.12 for ensuring and protecting the freshwater incomes from the wise use of wetlands and so conserving benefits provided to society, at the present and in the future (http://www.ramsar.org/sites/default/files/documents/library/cop12_res12_water_requirements_e.pdf). Despite the benefits of designation, there are gaps in the designation of internationally important wetlands, for example the Cerrado wetlands (*Veredas*) are vital for the regulation of water flows of most rivers in Brazil yet have not been recognized. Over the next 20 years, there will be unprecedented pressure on resources in the Americas as global demands for food, energy, and shelter increase. At the center of the crisis is water (UN, 2013).

2.3.3 Energy security

The UN defines energy security as being able to have access to clean, reliable and affordable energy, which is crucial for such human activities as cooking, heating, lighting, communications and production, but in addition we must consider the reliability and price of the energy source (IEA, 2017). From the perspective of sustainability, energy has been described as "the golden thread" connecting economic growth, social equity, and environmental sustainability. With this in mind, the UN General Assembly in 2012 embraced the 'Sustainable Energy for All' objectives for 2030, aiming to: 1) achieve universal access to modern energy, 2) double the historic rate of improvement of energy efficiency, and 3) double the share of renewable energy in the global energy mix. In 2015, SDG7 was adopted for 2030, to "ensure access to affordable, reliable, sustainable, and modern energy for all," building further on the three Sustainable Energy for All objectives. Later in 2015, at the historic 21st Conference of the Parties to the UN Framework Convention on Climate Change, countries from around the world committed to nationally determined contributions, many calling for progress on the sustainable energy agenda (IEA, 2017). As a result, SDG 7 is interconnected with all other SDGs.

In about 33% of the countries of the Americas, including the North American countries of Canada and USA, 100% of people have access to electric power, in the other 67% at least 80 % have access, with exception of Haiti where only 38% of the population receives electric power. The role biodiversity plays in providing energy through fuelwood for cooking, heating and lighting in localities with little or no access to electric power is enormous, although it is often poorly quantified. Hence, unsustainable use of fuelwood in such areas is a major threat to energy security, particularly for the already disadvantage people.

When discussing energy security in the context of NCP, it is important to realize that only part of the past, current and future energy supply comes from nature: other sources not considered in this framework, including fossil fuels and those renewables that are derived from solar and wind. Considering the fast changes in the energy sector, in terms of main energy sources, production and transportation structure and also demands, embedded in complex political settings and unequal distribution of different energy sources around the world, an assessment of NCP contribution to energy security is difficult. Overall energy security, including aspects related to energy equity and sustainability, is high for North American countries, intermediate for South American countries and low for Caribbean and Central American countries.

The World Energy Council (2017) considers Latin America, with its high dependence on hydropower for electricity production, as vulnerable to extreme climate events and facing socio-economic challenges from the impacts of large hydropower projects. The report also acknowledges the surprisingly low use of renewable energies based on sun and wind, despite high potential. Development of these techniques could not only be efficient in terms of energy security, but also to decrease pressures on NCP by reducing emissions of climate-relevant gases from fossil fuels and impact of hydropower or biomass extraction. In the USA, in contrast, electrical generation from non-hydro renewables (including solar, wind and biomass-based energy) has more than tripled over the past decade, surpassing in 2014 hydro-power electrical generation (EIA, 2017), thus reducing direct pressures on NCP. In many parts of the Americas, however we can expect conflicts between energy security and food security, in particular as fossil fuels will be replaced by fuels derived from biomass (Fischer et al. 2009, Koizumi 2014). This is also expressed in the values of the World Energy Council's Energy Trilemma Index that ranks countries based on the three dimensions energy security, enerqy equity (accessability and affordability) and environmental sustainability (Table 2.18).

Table 2.18. The Energy Trilemma Index is shown from countries in the Americas. This index includes energy security, energy equity and environmental sustainability to produce an overall score. Country rank indicates the position of each in the context of the global list, which goes from 1 (Switzerland) to 125 (Benin).

Country rank	Subregion/Country	Energy Security	Energy Equity	Environmental Sustainability
	Caribbean			
77	Dominican Republic	121	75	13
79	Panama	118	76	47
90	Trinidad & Tobago	99	48	123
98	Jamaica	120	89	72
	Central America			
42	Costa Rica	89	64	5
52	Mexico	59	71	55
71	El Salvador	87	83	21
110	Honduras	107	101	118
	North America			
14	USA	4	13	73
22	Canada	5	11	96
	South America			
27	Uruguay	40	51	16
38	Chile	44	66	48
41	Colombia	36	80	10
50	Ecuador	50	46	79
57	Brazil	68	70	46
58	Argentina	48	69	69
62	Venezuela	21	68	87
64	Peru	54	84	38
89	Paraguay	96	86	57
Source: World Energy Council available at: https://trilemma.worldenergy.org (accessed March 9, 2017).				

2.3.4 Health

Human health is a core component of quality of life, which has many measurable attributes (Salim et al., 1999). According to the World Health Organization, "health is a state of complete physical, mental and social well-being and not merely the absence of disease or infirmity," and also, healthy living conditions have physical, biological, cultural and spiritual components (Corvalán et al., 2005). Health is determined by social, economic and environmental factors, and biodiversity supports a diversity of NCP that are essential to human health and quality of life, including food (provide nutrition), medicinal organisms and their products, physical and psychological experiences, regulation of water quality, regulation of air quality, regulation of hazards and extreme events, and regulation of organisms detrimental to humans; thus biodiversity is a key determinant and the conservation and sustainable use of biodiversity can benefit human health (World Health Organization and Secretariat of the Convention on Biological Diversity, 2015).

Attaining good health and wellbeing is the aim of SDG3. In fact, health is intrinsically related to food and water security (sections 2.3.1 and 2.3.2).

Biodiversity plays a key role for a dietary balance. However, population increase, urbanization and industrial agriculture have changed production and consumption patterns. Calories obtained from meat, sugars and oils have increased during the last decades. In contrast, the consumption of fiber-rich foods such as whole grains, pulses and roots have declined (World Cancer Research Fund International, 2014). This nutrition transition affects dietary patterns in many countries of the Americas region, where the increase in consumption of meat and processed food has favored the occurrence of noncommunicable diseases (Webber et al., 2012; Claro et al., 2013; Pou et al., 2016).

Historically, biodiversity has sustained medicine around the world. Plants have been used for health purposes by indigenous people and local communities, as a result of practicing the traditional medicine, defined as "the sum total of the knowledge, skill, and practices based on the theories, beliefs, and experiences indigenous to different cultures, whether explicable or not, used in the maintenance of health as well as in the prevention, diagnosis, improvement or treatment of physical and mental illness" (World Health Organization, 2013). Nevertheless, medicinal plants are not only used by locals, but also for international trade to produce extracts, phytopharmaceuticals and cosmetics. It is estimated that the average global export of medicinal plants during the year 2014 was around 702,000 tons valued at $3.6 billion. Chile and Peru are important suppliers, while USA is the major consumer (Vasisht et al., 2016). The penicillins, as well as nine of the 13 other major classes of antibiotics in use, are derived from microorganisms, and more than half of the approved drugs by the USA Food and Drug Administration between 1981-2010 had natural product origins. This is in spite of the fact that only a small fraction of the total plant species that populate the earth have been studied for pharmacological purposes (World Health Organization & Secretariat of the Convention on Biological Diversity, 2015).

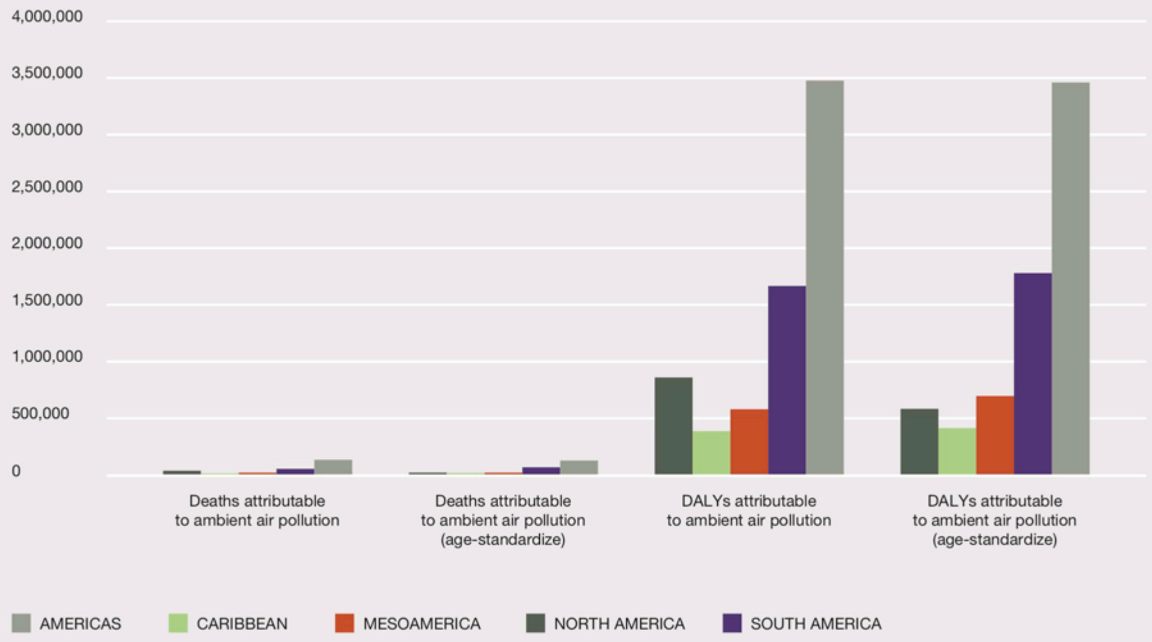
In North America, there is a robust body of literature regarding the health effects of access to nature for different social groups. Specifically, in urban areas, access to parks and green spaces improves health not only by the physical activities one can conduct in these places, but also simply providing views from windows and even indoor plants can produce a similar, positive health outcome (Grinde & Patil, 2009). In Vancouver, Canada, providing elderly adults (65-86 y.o.) access to nature was shown to provide greater mental, social and physical health. This was in response to therapeutic landscapes with "green" (i.e. vegetated) and "blue" (i.e. aquatic) features that, given the limited mobility of some elderly citizens, were accessed not only via direct interaction, but also via perception (i.e. looking from window) (Finlay et al., 2015). Similarly, inner-city hispanic youth in Houston, Texas (USA) were found to have improved health when there were larger and more abundant trees near them, as well as smaller distances between tree patches (Kim et al., 2016).In terms of regulation of organisms detrimental to humans, freshwater wetlands as riparian buffer may improve the bacterial water quality, by eliminating livestock manure in streams as well catching of bacteria by the riparian vegetation (Collins & Rutherford, 2004). Deforestation degrades the disease regulation services and may increase disease transmission such as with Dengue fever, yellow fever, leishmaniasis (Walsh, 1993; Willcox & Ellis, 2006) and malaria (Walsh, 1993; Vittor et al., 2006; Pattanayak & Yasuoka, 2008). Mining operations in Colombia have been shown to be reservoirs for malaria (Castellanos, 2016). Both selective logging and general deforestation may amplify other disease risks (Foley et al., 2007).

Beyond the direct impact of diseases on human health, forest degradation also impacts medicinal plant populations. Forest degradation and transformation negatively impacts the discovery of potential remedies for people in the developed world and also causes the erosion of one of today´s primary health care options for Amazonian's urban and rural citizens (Shanley & Luz, 2003). An increase in insect-vector diseases is also likely as hydroelectric dams proliferate on the Amazon and its tributaries, despite the fact that some consider hydropower a clean energy source. The necessary access to water and sanitation for good health is discussed in section 2.3.2. Nevertheless, an effort to assess the health and social status of indigenous and tribal peoples relative to benchmark populations from a sample of 23 countries (including five from South America, two from North America and one from Mesoamerica) provide evidence of poorer health and social outcomes for indigenous peoples than for non-indigenous populations (Anderson et al., 2016). The reduced access to land and its biodiversity, and consequent change in indigenous diet has contributed to this scenario.

Despite biodiversity's important role in regulating air quality, the complex mixture of emissions from industrial activity, households, cars and trucks have a harmful effect on health. In high-income countries, urban outdoor air pollution ranks in the top ten risk factors to health, and is the first environmental risk factor. Air pollution, natural disasters, disease outbreaks, environmental contaminants such as lead exposure, unsafe water and lack of sanitation, all contribute to the high percentage of deaths attributed to environmental causes (Figure 2.33).

Figure 2 33 **Deaths and number of disability-adjusted life years (DALYs) due to ambient air pollution in 2012.**

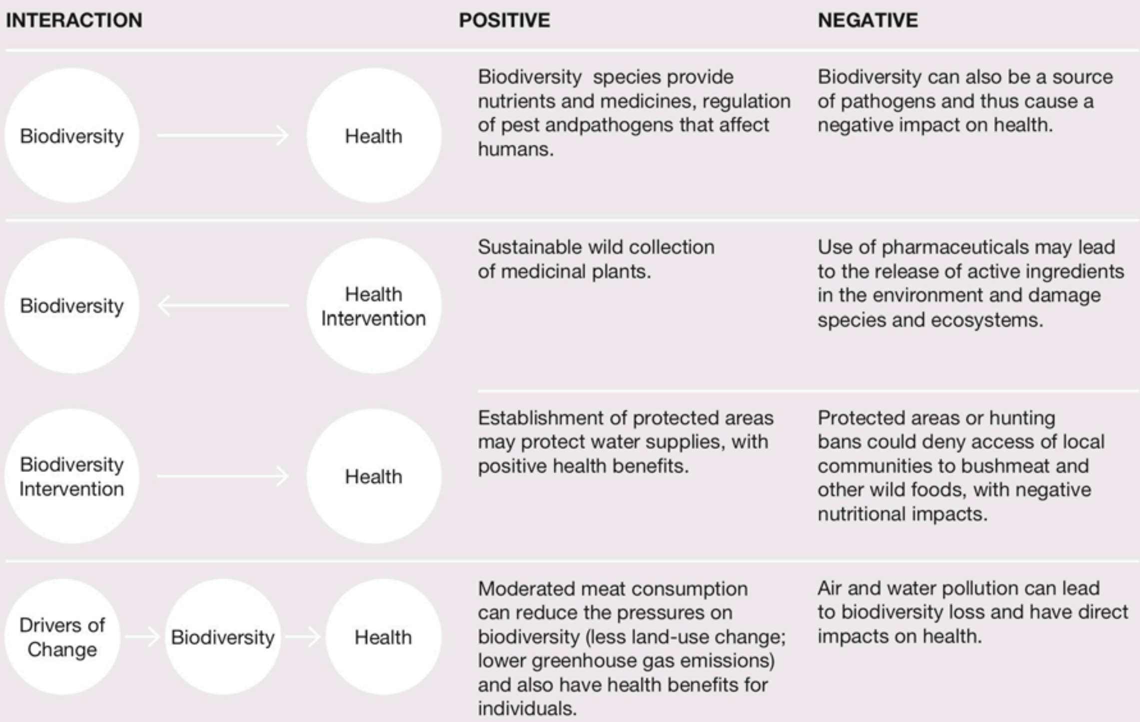
Source: World Health Organization (2017). Global Health Observatory data. http://www.who.int/gho/database/en/. Date accessed: April 23, 2017.



Coastal ecosystems can alleviate the impacts of an extreme event on human systems (Bravo de Guenni et al., 2009). For example, mangroves reduce the risk of wave damages, large storms, tsunami damage, erosion and bind soils together and keep up with the sea level rise (Spalding et al., 2014); forest and wetlands combat flash flooding, acting like sponges to absorb the excess of water after storms and releasing it more slowly (Delach, 2012); forests prevent landslides by reinforcing soil layers with roots and reduce soil moisture through interception, evaporation and transpiration (FAO, 2010), and the control of invasive species in forest could reduce the impact of destructive fires (Delach, 2012). Suitable management of ecosystems can be an important mechanism to reduce vulnerability and reduce negative impacts of extreme events.

Finally, direct drivers of biodiversity loss that affect human health include land-use change, overexploitation, habitat loss, pollution, invasive species and climate change (World Health Organization & Secretariat of the Convention on Biological Diversity, 2015). The largest health impacts due to biodiversity losses are projected to be increases in undernutrition, and higher rates of disease, injuries and deaths from natural disasters. The interaction between biodiversity and health are both positive and negative (Figure 2.34), resulting in trade-offs that will be critical for decision making.

Figure 2 34 A typology of biodiversity-health interactions. Source: Adapted from the World Health Organization and Secretariat of the Convention on Biological Diversity (2015).



2.3.5 Sustainable livelihood

Livelihoods depend upon economic conditions, such as employment and income, as well as broader socio-cultural aspects that affect “ways of living” and incorporate nature via cultural identity, sense of place, and social cohesion. As seen in section 2.2, numerous NCP directly support income security, but in different ways between subregions. For example, between 1995 and 2012, the number of people employed as commercial fishers and fish farmers declined by 15.4% in North America, but increased by 49.8% elsewhere in the Americas (FAO, 2014b), mostly associated with the rise of aquaculture rather than native fisheries. In the Caribbean, however, coastal ecosystems continue to support a fisheries industry, which contributes about $1.2 billion annually in export earnings (CARSEA, 2007). The importance of fisheries to livelihoods in the Caribbean is reflected not only in monetary figures; fish products account for ~7% of the protein consumed by people in the Caribbean subregion and is a “way of life” for fishermen, which transcends being a merely “job” or income. Furthermore, sustainable small-scale fishing livelihoods can be compatible with marine protected areas and in turn contribute to the implementation of these conservation initiatives (Charles et al., 2016). In this way, coastal ecosystems can be used for multiple benefits to multiple beneficiaries, providing food, income and livelihoods to fishermen, but also tourism and travel to other stakeholders, which in the case of the Caribbean constituted 15.5% of the subregion’s total employment in 2004 (nearly twice the global average). In this same time period, coastal tourism and travel in the Caribbean contributed $28.4 billion to the subregion’s GDP, which is 13% of total economic production (CARSEA, 2007).

Similarly, in the Andes mountains, sparsely-vegetated highland ecosystems support the livelihoods of about 6% of the biome’s 85 million human inhabitants, while another 34% of the population live off grazing lands often interspersed with other habitat types. Plus, in this area, approximately 5% of people live in and their livelihoods depend on protected areas (Huddleston et al., 2003). In the Andean altiplano (3,900 – 4,900 m.a.s.l), shared between Argentina, Bolivia, Chile, and Peru, various indigenous communities derive not only their economic security from the NCP of this biome, but their rich cultural tradition of beliefs and rituals, and a particular worldview that mediates their interaction with the environment, are also based on the particular elements that nature provides here (Lichtenstein & Vilá, 2003). Furthermore, secondary stakeholders depend upon these ecosystems for their livelihoods, given that mountain ecosystems contribute to the human populations at lower elevations via the regulation of water flow, energy, waste assimilation and drinking water (Bradley et al., 2006; Buytaert et al., 2006; Vuille et al., 2008).

For the temperate forests of North and South America, we can distinguish the contribution of nature to specific stakeholders, including direct users, such as many rural communities of both indigenous and immigrant ancestry, whose livelihoods depend on benefits from these forest ecosystems for material subsistence (e.g. logging, Nelson et al., 2008) or cultural practices (e.g. non-timber forest products, Ladio, 2011). Research in both North and South America has described how community-based restoration and management not only improve ecosystem services and benefits, but also can be part of work security and increasing social capital (e.g. Donoso et al., 2014). However, in the negotiation of trade-offs between livelihoods of primary and secondary beneficiaries, rural communities often comprise a smaller portion of the total population in the Americas, and most people reside in urban areas, indicating that decision-making power rests with “secondary users” who have a less direct relationship to NCP for their livelihoods.

In the Americas, we are confronted with major challenges in addressing sustainable livelihoods, in part because there are great disparities in economic security, which represents an obstacle in achieving SDG8 and SDG10. Income distribution both between and within subregions is very heterogeneous. For example, the mean per capita GDP is $50,935 (±$1,711 Standard Deviation) for countries in North America, $9,883 (±$5,965) in the Caribbean, $8,436 (±$4,585) in South America, and finally $5,477 (±$3,554) in Mesoamerica (IMF World Economic Outlook Database, 2015). As can be seen from the standard deviation, the between-country variability is high in all subregions except North America (i.e. Canada and USA). Furthermore, while from 1993 to 2013, Mesoamerica, the Caribbean and South America reduced their overall rates of moderate poverty ($2.5 to $4 per day) from 16.8% to 12.9% and extreme poverty (less than $2.5 per day) from 26.6% to 11.5%, at the same time 25-30 million people are still considered vulnerable to falling back into detrimental economic conditions (UNDP, 2016). In addition, both Canada and the USA have seen increases in income inequality, attaining levels of inequality that are higher than the Organization for Economic Cooperation and Development (OECD) average (OECD, 2008). For its part, Mesoamerica, the Caribbean and South America together also have 10 of the world’s 15 countries with the most unequal income distribution (UNDP, 2016).

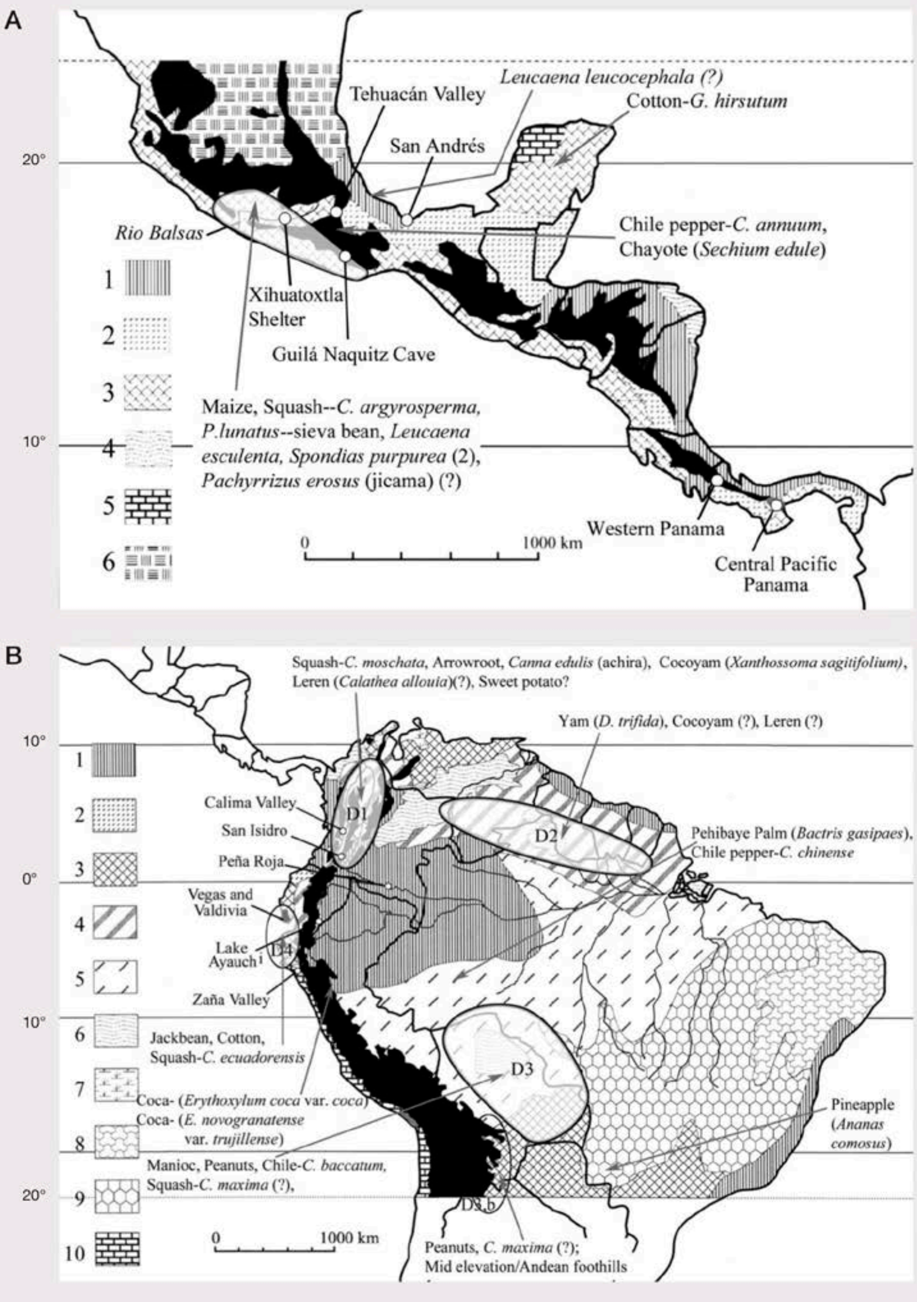
Yet, biodiversity-based livelihoods can be sustained when people have the social capital to cope with and recover from stresses and shocks, while maintaining or enhancing their capabilities and assets both now and in the future without undermining the natural resource base (Chambers & Conway, 1992). At the same time, though, broader policies and strategies are necessary, considering that sustainable livelihoods in today’s world depend on telecoupled and globalized processes. As such, increases in corporate social responsibility and environmental sustainability initiatives are important to harness market forces and orient them towards favoring desired outcomes like specific sustainable livelihoods (e.g. small-scale farming, well-managed fishing). Meanwhile, government strategies, such as multi-use protected areas, create the conditions to not only conserve nature, but protect the livelihoods that have evolved for millennia in these same ecosystems. Indeed, such multi-faceted strategies are requisite to achieve SDG14 (Life below water) and SDG15 (Life on land), which in turn are underlain by sustainable livelihoods.

2.4 Contributions of indigenous people and local communities to biodiversity and nature’s contributions people

Cultivated plants, or cultigens, are a main inheritance we receive from nature. This heritage contributed greatly to the development of mankind, and the history of cultigens is part of our own history as they were created by humans and have been used for millennia (Krapovickas, 2010). For example, maize is the cereal of the peoples and cultures of the Americas. Known or postulated geographic zones of domestication for some neotropical crops, on the basis of molecular, archaeological, and ecological evidence, show various origin areas (Piperno, 2011, Figure 2.35).

Figure 2 35 **Areas where various tropical crops in Central A and South America B are thought to have been domesticated.**

Open circles are archaeological and paleoecological sites with early domesticated crop remains. The numbers in parentheses after a taxon indicate that more than one independent domestication event occurred. The possible area of origin for the sieva bean extends into the Pacific lowlands north of the oval area. The oval here and those in B labeled D1–D4 designate areas where it appears that more than one or two important crops may have originated. Arrows point to approximate areas and are not meant to denote specific domestication locales. Modern vegetation zone guides are (a) 1, tropical evergreen forest; 2, tropical semievergreen forest; 3, tropical deciduous forest; 4, savanna; 5, low scrub/grass/desert; 6, mostly cactus scrub and desert; and (b) 1, tropical evergreen forest (TEF); 2, tropical semievergreen forest (TSEF); 3, tropical deciduous forest (TDF); 4, mixtures of TEF, TSEF, and TDF; 5, mainly semievergreen forest and drier types of evergreen forest; 6, savanna; 7, thorn scrub; 8, caatinga; 9, cerrado; 10, desert. Source: Piperno (2011).



The oldest civilizations of America - from the Olmecs and Teotihuacans in Mesoamerica, to the Incas and Quechuas in the Andes of South America - were accompanied in their development by potato plants

(Serratos Hernández, 2009). The first cultivated potatoes were probably selected between 6,000 and 10,000 years ago, north of Lake Titicaca in the Andes shared between Peru and Bolivia, and *Solanum* L. sect. *petota* grows from the southwest of the USA to the south of Chile (Rodríguez, 2010). Archaeological and genetic evidence has helped to understand the origin of three of the oldest and most important American crops in pre-Columbian and present times: maize, common bean and Lima bean (Chacón, 2009). Archeological evidence has also pointed out least 83 Amazonian native species containing populations domesticated to some degree before European conquest, indicating that Amazonia was also major center of crop domestication (Clement et al., 2015).

According to the Vavilov concept of plant origin centers, major food crops developed over millennia and originated from a central point from which humans dispersed them. These ‘‘centers of origin’’ represent locations with great genetic diversity of crop species (Hummer & Hancock, 2015). The Americas host a diverse and rich variety of species that have been cultivated by humans for food and a wide variety of resource uses. Cultures throughout the Americas have continually enriched world food and nutrition (Janick, 2013, Table 2.19).

Table 2.19. Selected indigenous crops of native plant species used throughout the Americas.

New World crops	Species	New World origin
Cereals and pseudocereals		
Amaranth	*Amaranthus* spp.	Mexico
Maize	*Zea mays*	Mesoamerica
Quinoa	*Chenopodium quinoa*	Andean highlands
Wild rice	*Zizania palustris*	Northern North America
Legumes		
Common bean	*Phaseolus vulgaris*	South America
Lima bean	*Phaseolus lunatus*	South America
Peanut	*Arachis hypogaea*	Bolivian-Brazilian-Paraguayan center
Cucurbits		
Chayote	*Sechium edule*	Mexico, Central America
Pumpkin	*Cucurbita maxima*	South America
Squash	*Cucurbita moschata, C. pepo*	Mexico
Solanaceous fruits		
Capsicum peppers	*Capsicum annuum, C. bacattum, C. chinenese, C. frutescens, C. pubescens*	South America, northern Peru, central Bolivia
Ground cherry, husk tomato	*Physalis peruviana, P. philadelphica*	Central America
Pepino	*Solanum muricatum*	Tropical America
Tomato	*Solanum lycopersicum*	Western South America
Roots and tubers		
Cassava	*Manihot utilissima*	Brazil
Potato	*Solanum tuberosum*	Peru and Bolivia
Sweetpotato	*Ipomoea batatas*	Central America
Fruits and nuts		
Annona	*Annona cherimola*	Brazil
Avocado	*Persea americana*	Mesoamerica
Black raspberry	*Rubus occidentalis*	North America
Brazil/Amazonian nut	*Bertholletia excelsa*	Amazon
Blueberry	*Vaccinium corymbosum*	North America
Cacao	*Theobroma cacao*	Tropical America
Cactus	*Opuntia ficus-indica*	Mexico
Cashew	*Anacardium esculenta*	Brazil

Cranberry	*Vaccinium macrocarpon*	North America
Guava	*Psidium guajava*	Tropical America
Jaboticaba	*Myrciaria cauliflora*	South America
Mamey	*Mammea americana*	West Indies, northern South America
Papaya	*Carica papaya*	Tropical America
Pejibaye palm	*Bactris gasipaes*	Southwestern Amazon
Pineapple	*Ananas comosus*	Tropical South America
Pitaya	*Stenocereus* spp.	Mexico
Strawberry	*Fragaria chiloensis*	Pacific coast: North and South America
Soursop	*Annona muricata*	Peru-Ecuador
Industrials		
Asai palm	*Euterpe oleracea, E. precatoria*	Amazon
Cotton	*Gossypium hirsutum, G. barbadense*	Central America, Brazil
Quinine	*Cinchona calisaya*	Peru-Bolivia
Rubber	*Hevea brasiliensis*	Amazon
Tobacco	*Nicotiana rustica, N. tabacum*	Mexico, Central America
Ornamentals		
Dahlia	*Dahlia* spp.	Mesoamerica
Fuchsia	*Fuchsia triphylla*	Hispaniola, South America
Petunia	*Petunia* spp.	South America
Sunflower	*Helianthus annuus*	North America
Source: Modified from Janick (2013).		

Over the centuries, indigenous management practices also shaped landscapes (Balèe, 2013 and contributed to highly productive soil formation, such as the dark soils in Amazonia (Schmidt et al., 2014). Modern tree communities in Amazonia are structured to an important extent by a long history of plant domestication by Amazonian peoples (Levis et al., 2017). A recent review on agrobiodiversity in the Amazonia pointed out that swidden agriculture may increase diversity in ecosystem ensuring *in situ* conservation. Another review confirms the importance of indigenous people and local communities for conserving and enhancing biodiversity in the Americas, and the important role of women in agrobiodiversity innovation, experimentation, selection and diffusion (Emperaire, 2017). Like their pre-Columbian ancestors, indigenous peoples and local communities are also contributing to high forest biodiversity in the Amazonia. As pointed out by Carneiro da Cunha & Morin de Lima (2017) “there is no clear-cut division between management of forests and agriculture as long as traditional long fallow systems endure. After all, fallows are intended to revert to forest, and to a large extent, it is fallow management that will result in humanised forests” Another example of agroforestry productive systems with edible plants is the *milpa* in Mexico, that is based on the culture around cocoa plantations in Tabasco or the culture around coffee plantations in Chiapas (González, 2004; De Beenhouwer et al., 2013; Cruz-Coutiño, 2014). Also significant are the systems based on the association of production of commercial cacao with wild species with the association of shade trees and nitrogen fixation in humid forests of Bolivia and Peru.

On-farm conservation of germplasm diversity is observed through great numbers of varieties of cultivated plants by indigenous people as local communities of mix-heritage people (Carneiro da Cunha & Morin de Lima, 2017). Table 2.20 shows the varieties cultivated for only one species (manioc) in different parts of Amazonia. According to Mendoza (2010), in Bolivia 19 wild species are widely represented, ranging from the lower Andean to Amazonian and Cerrado landscapes.

Table 2.20. Varietal diversity of Manioc (*Manihot esculenta*, Euphorbiaceae) in terms of taste in South America.

Indigenous Peoples & Local Communities	Location	Sweet	Bitter	Sweet + Bitter
Amuesha (Aruak)	Peru			204
Wanana, Tukano, Arapaso	Middle Uaupés, AM, Brasil			137
Pluri-ethnic communities: Barcelos	Middle Rio Negro, AM, Brazil			120
Piaroa (Piaroa-Saliban)	Cuao and Manapiare (Orinoco basin), Venezuela			113
Pluri-ethnic communities: Santa Isabel	Upper-Middle Rio Negro, AM, Brazil			106
Tukano (Uaupes)	Uaupés, AM, Brazil			100
Aguaruna (Jivaro)	North Central Peru			100
Huambisa (Jivaro)	Peru			100
Tatuyo (Tukano)	Uaupés, AM, Brazil			100
Wajãpi (Tupi-Guarani)	Amapá, Brazil	94	3	97
Aluku (“quilombola”)	French Guiana			90
Makushi (Karib) e Wapishana (Aruak)	Roraima, Brazil Guyana, Venezuela			76,77
Cubeo, Piratapuia e Tukano (Tukano), Tikuna (Tikuna) e Sateré-Mawé (Mawé)	Cuieiras river, Lower Rio Negro, AM, Brazil	65	5	70
Wayana (Karib)	French Guiana			65
Pluri-ethnic communities	Middle Rio Negro, AM, Brazil			64
Bare (Aruak)	Upper Rio Negro, AM, Brazil			60
Local communities Mamirauá and Amanã	Middle Solimões, AM, Brazil			54
Kayapo-Mebêngôkre (Jê)	Pará, Brazil			46
Kuikuro (Karib)	Upper Xingu, Mato Grosso, Brazil			36-46
Pataxó (Macro-Jê)	Bahia, Brazil			34
Paumari (Arawa)	Purus, AM, Brazil			14 - 30
Krahô (Timbira-Jê)	Tocantins, Brazil	9	12	21
Canela-Ramkokamekra (Timbira-Jê)	Maranhão, Brazil	7	9	16
Kaiabi (Tupi-Guarani)	Mato Grosso, Brazil	9	6	15
Enawenê-Nawê (Aruak)	Mato Grosso, Brazil	14	1	15
Source: compiled by Carneiro da Cunha & Morin de Lima (2017).				

Indigenous knowledge and management practices also play an important role in conserving aquatic resources in both freshwater and marine environments. The First Salmon ceremony practiced by many indigenous groups in the Pacific Northwest of North America is an example: it indicates the annual opening of the fishery and its ecological function is also consistent with cultural values encoded in stories and rituals about respecting salmon, allowing creatures to reproduce, not interfering with the leaders in migration, and reciprocal obligations of humans and non-human beings, in general (Berkes, 2008). For example in the Amazonian hydraulic system of embankments (locally known as *camellones*) in the Llanos de Moxos (Bolivia) and Acre and Rondonia (Brazil) has been a way of taming the landscape in the face of continuous floods, instead of taming species, designing a variety of permanent and seasonal habitats and ecotones for the fish fauna and other wild foods may proliferate; in addition to providing elevations for crops, housing and ponds to store water and food, such as mollusks, fish, reptiles, generating more attractive conditions for the fauna of interest (Denevan, 1966; Erickson, 2010). Also in the highlands of the Andes, the agricultural infrastructure of the *suka kollus* in relation to the Titicaca Lake (shared by Peru and Bolivia) is the oldest one in South America (Erickson, 2006). Basically, these structures consist of a series of land platforms surrounded by water channels and arranged in different ways according to the slope and are constructed by digging the ground for the formation of earth channels and the soil of the channels is distributed above the platforms, raising the original surface of the ground. The *suka kollus* form a microclimate that allows to obtain high yields (potatoes and fish farming), reduce the effects of frost on crops, recycle the nutrients contained in the organic matter of the canals, drain excess water and irrigate crops (Erickson, 2006).

Indigenous and local knowledge systems plays a key role in food production systems, they are important resource in the conservation of domestic crop varieties for many species and wild plant as well as animal communities (Nakashima & Roué, 2002). For example, traditional livestock production systems include animals as an integral part of the landscape; in some parts of the highlands of Peru alpaca and vicuña are considered flagship species, and this has allowed rural tourism (Hoffmann et al., 2014), while in Mesoamerica and South America, oxen or bulls have been used for plowing and horses or mules for cultivation (Starkey, 2010). However, in the conversion of natural ecosystems to those altered for intensive food production, many of the properties linked to cultural services of indigenous peoples are reduced, and one of the major problems indigenous peoples face is land use change and degradation, which leads to the transformation or loss of traditional knowledge (Danver, 2015). For example, the *aynuqas* system applied in the Andes of Bolivia and Peru, organizes the agricultural production to dry land as the livestock by the use of the grasses (Benavidez, 1999). It is part of the historical memory of the community and is one of the constituent elements of its identity; the succession and rotation of the plots function as a spatial and temporal reference that preserves the recollection of the crops, good or bad (Riviére, 1994). Traditional management has varied like the intensive use of fodder, the use of chemical fertilizers and pesticides, as well as incentives for raising cattle (Hervé & Ayangma, 2000).

2.5 Addressing access, benefit sharing and values

Access is fundamental to obtain the benefits that accrue to well-being from the direct use and experience of nature; benefit sharing too is required for nature’s contributions to people to be distributed among different stakeholders, particularly those that are not directly connected to an ecosystem and/or have less power. For example, family farmers account for more than 80% of all farmers in the Americas (excluding the Caribbean), which is below the global proportion of 98%. Nevertheless, only 18% of agricultural lands are held by family farmers in South America and 68% in North America and Mesoamerica (Graeub et al., 2016). Inequity in land access, particularly in South America, conditions the human-nature relationships that can take place in a given location, which in turn leads to different benefits but also affects the values that are represented in decision-making regarding how ecosystems are used towards quality of life. Consequently, specific policies have been developed to promote access and benefit sharing (see examples in Chapter 6), but while many nations enshrine these as rights, even delineated in their constitutions, legislation and jurisprudence (e.g. more than half of Mesoamerican and South American countries recognize a legal right to water (Mora Portuguez & Dubois Cisneros, 2015)), little information exists on the broad-scale status and trends of the relationship between access and benefit sharing, values and well-being, limiting comparisons between subregions or biomes.

Nonetheless, it is possible to find the consequences of a lack of access and benefit sharing when socio-environmental conflicts arise throughout the region, which are based not only on divergent uses and interests between social groups, but also on inherent differences in values and knowledge systems that are at play in these trade-offs (Temper et al., 2015). For example, conflicts over access and benefit sharing can occur due to discrepancies over resource use (e.g. mining versus water rights) or the distribution of costs/benefits (e.g. pesticides to increase crop yields versus health impacts to adjacent communities), but also broader social constructs based on worldviews regarding property tenure (e.g. indigenous versus private rights) and management jurisdictions (e.g. traditional or commons management versus state protected areas). The ecosystem services literature has emphasized payment for ecosystem services (Wunder, 2005, see Chapter 6), which is one attempt to re-connect human societies and relate diverse stakeholders to not only receive a benefit, such as food or water (FAO, 2011a), but also contribute to its continuation or compensate the stakeholders who provide the service. This approach has also been identified as a way to monetarize the contribution of protected areas to social well-being and better include them in decision-making, specifically in Latin America (FAO, 2009). For example, to better link the use and provision of water, in Quito, Ecuador, city dwellers pay small landowners in the headwaters of streams and rivers to conserve riparian vegetation (Espinosa, 2005). In 2013, Colombia also implemented a payment for ecosystem services initiative for water resources (Decree 953/13). Similarly, Costa Rica's national payment for ecosystem services scheme aims to conserve forests, and their associated ecosystem service, but while being lauded for its innovation, the program has also been criticized for not actually translating into economic benefits for participants (Arriagada et al., 2015). Therefore, while payment for ecosystem services has received significant attention from academics and governmental institutions, other proposals are also needed, including ecotourism development models, land use zoning and informed consent for development projects (see Chapter 6 for a full discussion on policy).

Nonetheless, conflicts regarding access and benefit sharing of nature appear to be increasing. Regional observatories have been established with a particular emphasis on mining (see https://www.ocmal.org/). Data from the Environmental Justice Atlas indicate that there are important differences in the number of conflicts per country and subregion (e.g. 137 in North America, 137 in Mesoamerica, 4 in the Caribbean, 490 in South America), which could be due to variation in the issues surrounding the use of nature, but it can also be the result of social factors, such as leadership, social capital, organization capacity and power relationships. Chile's National Human Rights Institute (http://www.indh.cl/mapadeconflictos) has shown that approximately 30% of socio-environmental conflicts in its jurisdiction occur in indigenous territories, but only 18% are caused by mining. Agro-industrial expansion has also been identified as an important driver of socio-environmental conflicts in subtropical and tropical portions of South America, where concentrated land tenure displaces local users and affects not only livelihoods (section 2.3.5), but also broader measures of biodiversity and ecosystem services (Cáceres et al., 2015). Some state protected areas can also have a similar effect to limit access and benefit sharing for local stakeholders who conducted traditional management of an area for hunting, fishing or swidden agriculture, giving rise to the term "conservation refugees" that are particularly problematic in areas like the Atlantic forest biome of Brazil (e.g. Bahia et al. 2014, see Chapter 1, section 1.6.2).

In conclusion, while payment for ecosystem services is one of the principal mechanisms proposed to help achieve Aichi target 3 and SDG10 and reconcile the distribution of the costs and benefits of providing nature's contributions to people, it does not necessarily account for inter-generational equity, nor does it account for alternative values and value systems. Since the current use of forests, fisheries, freshwater and other natural resources in most parts of the world is judged to be unsustainable and species loss to extinction has accelerated (Travis & Hester, 1991; Cohen, 1995; Jackson et al., 2001; Brown et al., 2005; MA, 2005; IPCC, 2007; Dawson et al., 2011, see also Chapter 3), the options and actions of future generations and stakeholders with alternative worldviews are compromised unless significant efforts are made to not only maintain and/or restore the capacity of nature to provide benefits to people, but also reconcile the access and benefit sharing of nature between social groups and generations (section 2.6 on Ecological footprint and biocapacity section). Such efforts require institutional change at several levels (see Box 2.6).

Box 2.6. Institutions mediating access to nature´s contributions to people.

Institutions link changes in the production of ecosystem services to changes in human well-being. Berbes-Blazquez et al. (2016) define institutions as “the arrangements that people design to regulate their interactions with ecosystems and may include organizations as well as rule systems”. While studies have assessed the value of nature to people and the effects of nature’s services on quality of life, scholarship has less frequently focused on the formal and informal institutional systems that determine the type of access members of a community have to nature’s contributions. Berbes-Blazquez et al. (2016) identify three specific gaps in knowledge in this area: 1) data concerning the effects of improved ecosystem service flows on human well-being, when power dynamics impact the distribution of benefits; 2) data concerning the co-production of ecosystem services, which involves a relationship between social and ecological systems; and 3) data concerning the historical factors that have shaped power relations between institutions and social groups that use and distribute ecosystem services.

Power dynamics between institutional and governance systems and various social groups are in large part responsible for shaping the way nature’s contributions to people are conceived and valued and subsequently produced and distributed. Understanding and changing such dynamics is particularly relevant to attain SDG10 (Reduced inqualities). At times, power dynamics leads to unequal access to nature’s services. For example, Costa Rica’s Limón Province produced $822 million of foreign exchange in bananas (a provisioning ecosystem service) and yet is ranked among the poorest provinces in the country (Sánchez Rojas et al., 2013). In this case, nature’s benefits became commodities in a market economy that contribute to foreign actors within powerful institutions at a cost to the quality of life of the community that produced and exported the goods (Berbes-Blazquez et al., 2016). This case also illustrates that local communities, which are most impacted by the degradation of nature and its services, often are also least able to advocate for themselves because external institutions exert a disproportionate amount of control over local management decisions.

Institutional relations also impact access to other services. Protected areas were first established in the USA at the end of the nineteenth century by way of the national park system. As subsequent decades saw the creation of protected areas across the globe, the social impacts of some became evident (Adams & Hutton, 2007). For example, the largest protected area in Central America, ‘Bosawas’ National Natural Resource Reserve in Nicaragua, was created in 1991 without the consultation of the indigenous communities and mestizo farmers inhabiting the area (Kaimowitz et al., 2003). NCP in this reserve are numerous: from land for agricultural production of corn, beans and rice, to coveted species of trees such as mahogany and cedar, to countless animals, all of which constitute the full range of services from provisioning to cultural. In the interest of conservation, the creation of protected areass has facilitated institutional actors to make rules about the use of nature’s contributions to people. With the establishment of protected areass as an institutional way to conserve nature and its benefits, it is also essential to understand the issues that arise with regards to access and sharing of those benefits.

Institutions regulate the control and access to ecosystem services. Well-functioning institutions may contribute to making ecosystem services become ecosystem benefits to all members of a community. However, there is a severe lack of empirical data on the accessibility of nature’s contributions to various social groups, as regulated by institutions. This means there is a lack of understanding about how power relations, values and knowledge systems vary between social groups and how these affect the institutions that shape environmental outcomes and access to benefits.

2.5.1 Nature’s contributions to people valuations

As noted in section 2.1, IPBES understands the multiple values and valuation methods involved in assessing NCP. This value plurality has been demonstrated throughout Chapter 2 with numerous quantitative and qualitative data. In Table 2.21, we synthesize examples of valuation from the perspectives of biophysical, health, socio-cultural and holistic ILK approaches. Then, we highlight the economic values of nature in the subsequent section and tables (see below). Specifically, the summary (Table 2.21) illustrates how a specific NCP can support different aspects and dimensions of good quality of life for humans. Indeed, it is crucial to understand that while the provision of ecosystem services depends upon their biophysical elements and dynamics (e.g. biodiversity, ecosystem functions), the translation of these ecological features into human well-being requires each NCP to be understood in terms of differential values and value systems.

Table 2.21. Nature's contributions to people support human well-being via multiple values and value systems.

	Valuation approaches			
NCP	**Biophysical**	**Health**	**Socio-cultural**	**Holistic ILK**
Food and feed	Edible plant and animal species (both domestic and wild) can be evaluated regarding such attributes as species richness or the surface area they cover. For example, section 2.2.1 highlights the extensive increases in monocultured commodity and industrial -scale crop and livestock species (e.g. corn, soybean, cotton, cattle) throughout the Americas. Also, in the Americas, though, we find traditional management practices that promote agricultural biodiversity (2.4) and use wildlife and fisheries for both food and feed (2.2.1, 2.3.1).	Food and feed clearly constitute a basic human need for good health, and significant improvements are observed in the Americas regarding overcoming malnutrition based on increased food availability (2.3.1). However, efforts are also increasing to enhance food production without pesticides (e.g. organic farming, sustainability certifications) to reduce health risks associated with industrial-scale food production (2.2.1).	Food-related activities (e.g. farming, livestock management, hunting, fishing) are closely related to regional cultures and ways of life (2.3.5). For example, North American cowboys Mesoamerican *vaqueros* and South American *gauchos* all work with and produce livestock, but these terms also represent regional identities reflected in music, culinary customs and handicraft (2.2.6).	In indigenous and local communities, mixed economies of cash and subsistence depend not only on the availability of local resources, but also on cultural knowledge regarding the ways of preparing, storing and distributing food (2.2.1, 2.2.6). For example, within the Inuit knowledge/value system, hunted animals (e.g. seals, polar bears) and humans are linked together in a spiritual relationship that both depend upon. Among the Quileute, this physical-spiritual connection with food is acknowledged by throwing the bones and head of the first salmon caught back into the river to ensure the salmon spirits' good will. Plus, ILK values systems often incorporate limits (e.g. taboos about when food items are temporally or spatially restricted) that promote conservation and protection of some species. Similarly, in the Bolivian Andes, ancestral agriculture and llama herding emphasizes respectful use of the environment linked to Mother Earth (*Pachamama*) (2.2.6, 2.4).
Materials and assistance	Biodiversity used for fiber and other materials include extensive lands used for forestry and certain crops, like cotton and flax (2.2.2).	Materials derived from nature provide shelter and clothing, which in turn are fundamental for a healthy life (2.2.2).	As with food and feed, certain livelihoods are associated with the production of this NCP (e.g. loggers), but materials derived from nature also constitute elements used in cultural practices (2.3.5).	Many of the health and socio-cultural benefits related to materials and assistance provided by nature is derived via the knowledge systems that allow their use and incorporation (e.g. methods of construction or tools and

				elements produced from natural elements) (2.4).
Energy	Biodiversity-derived energy includes the species and biomass produced specifically for this purpose (e.g. biofuels) and also secondary products of other activities (e.g. left over forestry biomass). Also, hydropower depends on biophysical dynamics of the watersheds related to hydrological regime (2.2.3).	Energy is a crucial element for human health (e.g. in colder climates that require heating), but pollution derived from energy use and production (e.g. air pollution, water contamination, ecosystem conversion for monoculture biofuels) is also an important driver of human health problems (2.2.3, 2.3.3).		
Medical, biochemical and genetic resources	Plant and animals species found in both domestic and wild settings are the source of numerous medical biochemical and genetic resources (2.2.4). For example, in the Brazilian state of Minas Gerais, 264 different plants are known to have medicinal properties, and 40% of them wild (Box 2.3).	Throughout the Americas, medicinal plants and animals can be meaningful contributions to human health (2.2.4, 2.3.4). For example, in the Brazilian Cerrado savanna, a diversity of medicinal plants is a significant component of the treatment that rural peoples can access for their own medical care (Box 2.3).	From this same example in Brazil, local healers –mainly women – are known as *raizeiras*, specializing in the use of these contributions from nature as part of their local identity and culture (Box 2.3).	Use of medical plants is intrinsically linked to indigenous and local knowledge and people who use and integrate them into their lives (2.2.4).
Learning and inspiration		Benefits of the learning and inspiration derived nature are well established regarding their effects on mental and physical health (2.2.5).	Many social, cultural and economic practices, (e.g. outdoor recreation) provide spiritual regeneration and leisure possibilities (2.2.5, 2.3.5).	Nature's relevance for spiritual practices is clear for many indigenous and local peoples. For example, hunting and wildlife are integral to indigenous cultures and their continuity, and the antiquity of this relationship is demonstrated from the depiction of wildlife and hunting in artwork from pre-Colombian ceramics and petroglyphs. Plus, nature-based rituals and dances still accompany indigenous persons from birth to death (2.2.6).
Supporting identities		The same elements of nature that are part of one's identity also directly affect health. For example, in Canada, loss of cultural identity	Nature-based elements of cultural identity can be found throughout the Americas. For example, in Brazil, over 400 sacred natural sites	Some indigenous peoples use clan names, construct totems or have relatives from other species, and therefore a species' extinction means

		is associated with negative mental health consequences for First Nations peoples, leading to high rates of depression, alcoholism, suicide, and violence with the greatest impact on youth (2.2.6).	are found in a variety of natural environments (e.g. streams, forest, coastal habitats) and are associated with a diversity of cultures and religions (2.2.6).	the loss of cultural identity, as well, based on a familiar understanding of the relationship between humans and other elements of biodiversity (2.2.1, 2.2.6).
Physical and psychological experiences		A systematic literature review showed that conclusive evidence supports that knowing and experiencing nature makes us generally happier, healthier people (see Russell et al. 2013) (2.1.1, 2.2.7, 2.3.4)	Experiences in nature are the basis for nature tourism in many regions of the Americas (expand, give examples), including visits to protected areas and coral reefs, snow skiing and birdwatching (2.2.7).	Physical and psychological experiences with nature form an important part of ILK systems. In some indigenous communities, for example, hunting prowess is a sign of leadership potential, and sharing hunting gains with family and others gives a good measure of community standing and self-esteem (2.2.6).
Maintenance of options	Protected areas are one biophysical measurement of maintaining the options of nature. These, in turn, can be evaluated in terms of terms of their extent or connectivity (2.2.8).	Preserving natural areas also allows experiences in nature that provide physical and mental health and happiness (see above).	The maintenance of options for nature permits specific cultural and social activities that allow relational values, which are important for both individuals and also social groups (2.2.8).	Preserving natural areas and the species they contain often means preserving the cultural context of indigenous people (2.4).
Climate regulation	Species distributions -and the use species in agriculture- is linked to climatic factors that are a principal component of the ecological niche of all species. Plus, changes in climate affect such biophysical measures as sea level, weather patterns and freshwater distribution and dynamics (2.2.9).	A change in climate has led to range shifts of disease carrying organisms, potentially introducing vector-borne diseases to new regions, with increased incidence of, for example, malaria and zika virus (2.2.9). Climate change also poses a threat to food and water security that ultimately affect health (2.3.1, 2.3.2).	Extreme events can cause not only death and incur large economic costs, but also drive human migration (‘climate refugees’) that affect regional and global demography and culture. In cities, improvements in design and planning can create microclimates can mitigate heat waves or heat islands (2.2.9).	
Regulation of freshwater quantity, flow and timing	Aquatic (rivers, streams, lakes) and terrestrial/aquatic (watersheds, wetlands) ecosystems, as well as their biodiversity, are crucial for	Access to freshwater in sufficient quantity is essential for human health, and currently in the Americas there is not only water scarcity in arid zones, but per	Lakes and rivers allow for many cultural and recreational activities and are the location for significant relational values inherent in the places where humans interact (2.2.10).	Damming of rivers in the Pacific Northwest of North America affected salmon harvests, which are important to First Nations peoples. Accordingly, their cultural knowledge includes bio-specific and local bio indicators which are

	controlling the dynamics of water cycles and regimes (2.2.10).	capita availability is also decreasing (2.3.2).		interpreted in specific management rituals and other activities regarding water regulation (2.4).
Regulation of freshwater and coastal water quality	Water quality depends heavily on maintaining intact biodiversity and ecosystems (2.2.11).	Contaminated water is associated with vector-borne diseases (2.3.4). For example, diahera cuased poor water sanitation is the cause of 1% (North America) 5% (Mesoamerica) 2% (Caribbean) and 3% (South America) of childhood deaths (<5 years) (2.2.11).		
Regulation of hazards and extreme events	Hazards and extreme events can be studied from a biophysical perspective, such as effects to biodiversity or changes in geomorphology as a result of erosion or landslides (2.2.12).	In its worst form, natural hazards (e.g. earthquakes) and extreme events (e.g. hurricanes) can cause human death (2.2.12).	Ultimately, natural disasters affect human livelihoods via impacts to economies and entire societies (2.2.12).	Traditional practices of subsistence harvesting of mangrove ecosystems have maintained vegetation cover and protected islands and mainland from storm surge and erosion (2.2.12).
Habitat creation and maintenance	Habitat creation and maintenance can produce higher connectivity, and enhance other NCP, such as pollination, pest control, water provision, and erosion prevention. In cities, improvement of microclimatic and hydrological conditions by presence of more and high-quality green infrastructure (2.2.12, 2.2.13).	The high importance of green spaces for improvement of living conditions and quality of live in city is especially important (2.2.12, 2.2.13).	Urban green spaces also constitute places that affect not just the health of inhabitants, but also provide the location of important cultural activities that constitute the relational values of nature (2.2.5).	Some traditional agriculture practiced by indigenous people is based on a rotation system of multi-aged and multi-species farm plots, which can increase diversity and create productive successional stage (2.4).
Regulation of air quality	The constituents and dynamics of air quality can be studied regarding their chemical and physical properties (2.2.14).	Air quality is a key factor that determines healthy environments. In 2012, there were more than 5 million deaths and disability-adjusted life years in the Americas (2.3.4).		
Regulation of organisms detrimental to humans	Harmful species, including viruses, plants and animals (both native and exotic) constitute an element of biodiversity in	The reduction of organisms that are detrimental to human health, especially tropical diseases (e.g. malaria, dengue, zika) has		Indigenous and local knowledge includes organic compounds that were used as poisons by some indigenous peoples, especially on arrows and

	themselves that can be studied as such (2.2.15).	immediate benefits for human health (2.2.15, 2.3.4).		spears, and in food. Some of these compounds are used today in insecticides (2.2.15).
Pollination and dispersal of seeds and other propagules	Pollination and seed dispersal are important ecosystem functions and the species that conduct these processes are part of the region's biodiversity. For example, there may be upwards of 500 species of native bees in Bolivia (2.2.16).	Pollination and seed dispersal are crucial for food production, which directly affect health and nutrition (see above Food and feed NCP).	Beekeeping not only provides pollination and food, but also constitutes an economic activity and livelihood for people throughout the Americas (2.2.16).	Indigenous local knowledge is rich in detail about local native bee species (Box 2.5).
Regulation of ocean acidification	Ocean acidification has been shown to have severe negative effects on ecosystem processes and on a large number of marine organisms (2.2.17).	Due to ocean acidification, marine fisheries can be negatively impacted, which could ultimately affect food that sustains healthy human populations (2.2.17).	Furthermore, the loss of fisheries also constitutes the loss of the fishing way of living (2.2.17).	
Formation, protection & decontamination of soils & sediments	Protection of soils from erosion and pollution is essential to prevent ecosystem degradation (2.2.18).	Good soil and sediment conditions clearly underpin other NCP, like Food and feed, that affect human health (see above).	Likewise, this NCP affects livelihoods based on nature, such as farming, forestry, livestock and also affects access to nature for cultural practices.	Indigenous management practices have shaped landscapes and contributed to highly productive soil formation, such as the dark soils in Amazonia (2.4).

At the same time, the monetary economic values of nature are especially important and can be directly incorporated into national budgeting and accounting procedures to rationalize cost-benefit analyses and planning. The ecosystem services monetary value in millions of USA dollars per year for the 33 countries of Latin America and the Caribbean is presented in Table 2.22. These estimates were based on data from Costanza et al. (2014), which updated the seminal Costanza et al. (1997) study. Furthermore, for Table 2.22, we incorporated data on Canada and the USA from Kubiszewski et al. (2017). Based on these studies, the total terrestrial ecosystem services monetary value for the Americas region was $24.3 trillion per year in 2011, which is equivalent to the region's 2011 GDP ($25.3 trillion per year, The World Bank Database, 2017a, accessed November 15, 2017).

Table 2.22. Monetary valuation of ecosystem services in the Americas.

Country	Ecosystem Service Monetary Value		
Subregion	**total US $ (x 1 million) / yr**	**US $ / ha / yr**	**US $ per capita / yr**
Canada	$3,584,661	$3,590	$99,985
USA	$5,331,051	$5,422	$16,586
North America	**$8,915,712**	**$4,056**	**$24,951**
Belize	$11,647	$5,070	$32,442
Costa Rica	$42,444	$8,306	$8,828
El Salvador	$14,953	$7,107	$2,441
Guatemala	$58,364	$5,355	$3,571
Honduras	$66,954	$5,952	$8,292
Mexico	$848,935	$4,322	$6,684
Nicaragua	$87,309	$6,697	$14,355
Panama	$51,622	$6,845	$13,139
Mesoamerica	**$1,182,228**	**$4,754**	**$6,844**
Antigua and Barbuda	$985	$22,378	$10,703
The Bahamas	$28,623	$20,622	$73,771
Barbados	$322	$7,495	$1,135
Cuba	$68,757	$6,257	$6,037
Dominica	$586	$7,815	$8,029
Dominican Republic	$26,451	$5,435	$2,512
Grenada	$289	$8,252	$2,699
Haiti	$15,837	$5,707	$1,479
Jamaica	$6,156	$5,601	$2,258
St. Kitts and Nevis	$201	$7,734	$3,591
St. Lucia	$537	$8,667	$2,905
St. Vincent & the Grenadines	$692	$17,755	$6,353
Trinidad & Tobago	$6,016	$11,728	$4,424
Caribbean	**$155,453**	**$7,081**	**$4,090**
Argentina	$2,212,877	$7,926	$50,969
Bolivia	$1,294,751	$11,786	$120,723
Brazil	$6,768,369	$7,948	$32,564
Chile	$298,938	$3,954	$16,656
Colombia	$717,015	$6,280	$14,867
Ecuador	$160,915	$6,277	$9,967
Guyana	$182,562	$8,492	$238,021
Paraguay	$496,869	$12,216	$74,841
Peru	$922,717	$7,179	$29,407
Surinam	$141,562	$8,641	$260,703
Uruguay	$125,929	$7,146	$36,693
Venezuela	$691,372	$7,580	$22,225
South America	**$14,013,877**	**$7,872**	**$33,492**
Americas Total	**$24,267,270**	**$5,711**	**$24,599**

Source: Total country-level values prepared by M. Hernández-Blanco from data in Costanza *et al*. (2014) and Kubiszewski *et al.* (2017). Greenland and French Guyana are not included.

Economic valuation of natural capital in this case was made using the benefit transfer methodology, which represents a first approach in estimating the monetary value of ecosystem services, especially when the area of scope is as large as an entire region. Further work must be developed to refine these findings. In addition, it is worth noting that the monetary value of each biome assessed here depends on the availability of research, and some are much more studied than others. Likewise, some ecosystem services have been more investigated than others, and therefore sometimes these can represent a significant portion of the ecosystem service monetary valuation in such summary exercises.

Nonetheless, from these data, we see that Brazil has the largest monetary value for its ecosystem services at $6.8 trillion per year, due to its size and the vast cover of its rainforest biome. The USA and Canada followed with $5.3 and $3.6 trillion per year, respectively (Table 2.22). Yet, when the monetary value of ecosystem services was assessed on a per unit basis, a different vision emerges. For example, per hectare, the highest values are found in the Caribbean, where countries like The Bahamas and Antigua & Barbuda have > $20,000 per hectare per year. In South America, both Bolivia and Paraguay also have very high values, at > $10,000 per hectare per year. Meanwhile, when expressed on a per person basis, Guyana ($238,021 per capita per year) and Suriname ($260,703 per capita per year) are about 10-fold greater than the average regional value of $24,599 per capita per year. Other countries that are at least 2-fold greater than the regional value include Argentina ($50,969 per capita per year), Paraguay ($74,941 per capita per year), Canada ($99,985 per capita per year) and Bolivia ($120,723 per capita per year). Furthermore, the economic contribution of nature varies by biome. Using the Ecosystem Services Partnership's database (Van der Ploeg & de Groot, 2010), we see that in particular coral reefs and wetlands are highly valuable in economic terms (Table 2.23). Kubiszewski et al. (2017) have shown globally that coral reefs and wetlands (coastal and inland) have extremely high economic value per hectare ($352,249 / ha, $140,174 / ha, respectively), while among terrestrial biomes, tropical forests are highest ($5,382.00 / ha) for natural environments and urban areas ($6,661.00 / ha) for anthropogenic habitats.

Table 2.23. Literature review of the habitat-specific monetary value of ecosystem services calculated for biomes in the Americas.

Biome	Number of studies	Range (US $ / ha / yr)	Median (US $ / ha / yr)
Boreal/Temperate Forests	7	0.01 – 4,400.00	82.72
Coastal/Coastal Wetlands	14	25.00 – 2,243.47	423.95
Coral Reefs	48	2.13 – 955,419.00	1,789.89
Grasslands	8	83.22 – 8,483.59	157.00
Inland Wetlands	2	83.22 – 8,483.59	--
Tropical Forests	25	0.60 – 1,627.50	24.27
Source: Data from Van der Ploeg & de Groot (2010,) and selected based on studies since 1997 that expressed their results in US $ / ha / yr. The range in monetary values is wide due to the diversity of ecosystem services evaluated in these studies, which also heavily affects the median of some biomes, especially tropical forests.			

2.6 Ecological footprint and biocapacity

Ecological footprint accounting assesses how much humans are demanding from the planet (i.e. ecological footprint), compared to what the planet's ecosystems are capable of renewing (i.e. biocapacity). Many human activities place demands on the planet's biodiversity and ecosystems (e.g. food production, housing and infrastructure, transportation). All of these demands compete for biologically productive space. Therefore, both demand on and availability of regenerative capacity can be approximated by adding up the mutually exclusive biologically-productive areas for providing these benefits of nature. By comparing the amount of capacity demanded for human uses with the amount of total biocapacity available, ecological footprint accounting measures the extent to which human demands on nature exceed the biosphere's capacity to meet those demands. If human societies take more than what nature can renew, then biodiversity and ecosystems services inherently will be put under stresses that threaten their continuity and ability to contribute to human quality of life in the future.

By 2003, the global ecological footprint exceeded the Earth's biocapacity by over 25% (Loh & Goldfinger, 2006). In 2012, demand exceeded capacity by 60%, and projections are that by 2020 it will be 75% greater than the planet's ability to sustain these uses (WWF, 2016). The status and trends of the Americas' ecological footprint coincide with this overall global increase (Table 2.24), which is occurring at the same time as observed declines in biodiversity (Wilson, 1988; Botkin et al., 2007) and decreases in the provision of many ecosystem services (MA, 2005, sections 2.2.1 to 2.2.18). Based on Global Footprint Network data, the Americas represent 22.8% of the total global ecological footprint, but only have about 13% of its human population. This higher than average resource use also is reflected in the Americas' per capita ecological footprint, which is 169% higher than the global average. Since 1960, all subregions in the Americas have experienced increases in their ecological footprint, with declines in the per capita biocapacity during this same time period. Nonetheless, the Americas hosts a great wealth of natural resources compared to the rest of the planet, as evidenced by the fact that the region contributes 40.5% of the world's biocapacity, and has 299% more resources available from nature per capita than an average global citizen (Table 2.24).

Intra-regionally, though, there are large variations in both ecological footprint and biocapacity. For example, North America has a 2.7, 4.1 and 4.6 times greater per capita ecological footprint than South America, Mesoamerica and the Caribbean, respectively (Table 2.24). At the subregional level, only South America retains a "reserve" of biocapacity for future use, due to its relative low ecological footprint and extremely high biocapacity; the other three subregions currently are exceeding nature's ability to renew the resources and services that contribute to human well-being.

Table 2.24. Overall, the Americas region has a high ecological footprint and biocapacity with large variation between subregions. Ecological Footprint Network data from 2012 are shown as global hectares per capita and the total number of global hectares. Negative values are presented in (red).

	Ecological footprint (gl ha per capita)	Biocapacity (gl ha per capita)	Reserve (gl ha per capita)	Ecological footprint (gl ha)	Biocapacity (gl ha)	Reserve (gl ha)
North America	8.2	5.0	(3.2)	2894.5	1751.6	(1,142.9)
Mesoamerica	2.7	1.3	(1.3)	436.7	218.3	(218.4)
Caribbean	1.8	0.7	(1.2)	69.5	25.0	(44.5)
South America	3.0	7.4	4.4	1195.2	2969.1	1,773.8
Americas Region	4.8	5.2	0.4	4595.9	4964.0	368.0
Global	2.8	1.7	(1.1)	20114.4	12243.5	(7,870.9)
Americas as % of Global	169.0	299.4		22.8	40.5	
Source: Global Footprint Network (2016) and see also WWF (2016).						

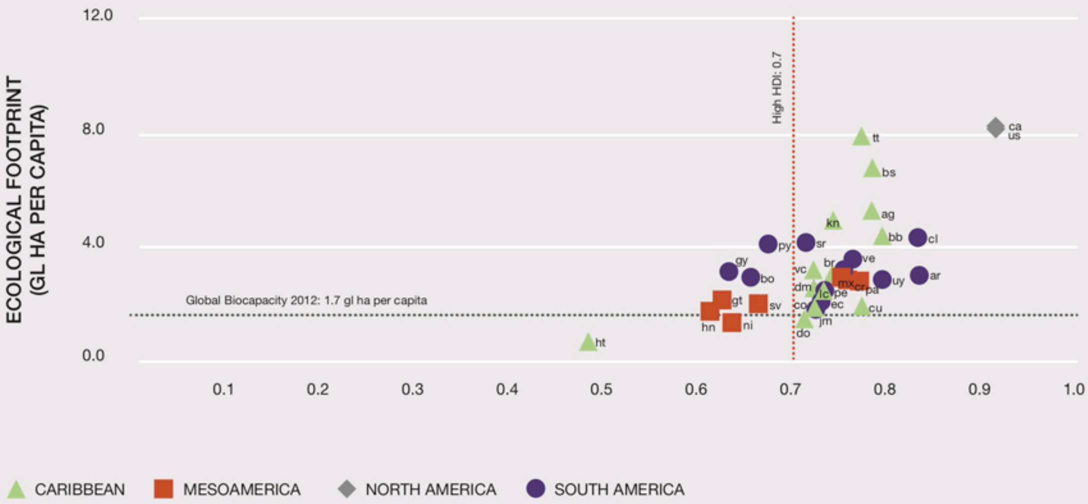
Based on the world's overall biocapacity to produce 1.7 global hectares per person in 2012, only four countries in the Americas are consuming (i.e. their ecological footprint) within these sustainability limits: Haiti, Dominican Republic, Honduras and Nicaragua. However, only the Dominican Republic is considered to have both a sustainable ecological footprint and a high HDI (Human Development Index). Indeed, the relationship between the consumption of natural resources and HDI is not uniform. For example, in the Caribbean, countries attain similar development outcomes (i.e. low variation in the x-axis for HDI), but have extremely different ecological footprints (i.e. high variation along the y-axis for global hectares consumed per capita). In contrast, in South America, countries attained very different development outcomes (i.e.

high variation in the x-axis for HDI) with similar ecological footprints (i.e. low variation along the y-axis for global hectares consumed per capita) (Figure 2.36).

In conclusion, most countries in the Americas are exceeding their biocapacity, and the fact that local environments are increasingly teleconnected to other parts of the planet means that biocapacity in one region or subregion may be used by beneficiaries in another. However, these findings also indicate that the relationship between consumptive uses of nature and development is not linear (i.e. high ecological footprints within a subregion do not always lead to increases in HDI). Consequently, policy-makers have an opportunity to implement strategies that reconcile sustainable use and human development (see Chapter 6).

Figure 2 36 **Ecological footprint as a function of UNDP Human Development Index per country.**

Source: All data are from the Global Footprint Network (2016) prepared by the Task Group on Indicators and the Knowledge and Data Technical Support Unit and WWF (2016). Countries included: Caribbean: Antigua & Barbuda (ag), Bahamas (bs), Barbados (bb), Cuba (cu), Dominica (dm), Dominican Republic (do), Grenada (gd), Haiti (ht), Jamaica (jm), Saint Kitts & Nevis (kn), Saint Lucia (lc), Saint Vincent & Grenadines (vc), Trinidad & Tobago (tt); Mesoamerica: Costa Rica (cr), El Salvador (sv), Guatemala (gt), Honduras (hn), Mexico (mx), Nicaragua (ni), Panama (pa): North America: Canada (ca), USA (us); South America: Argentina (ar), Bolivia (bo), Brazil (br), Chile (cl), Colombia (co), Ecuador (ec), Guyana (gy), Paraguay (py), Peru (pe), Suriname (sr), Uruguay (uy), Venezuela (ve). Aruba, British Virgin Islands, Cayman Islands, Guadeloupe, Martinique, Montserrat and French Guiana are not represented due to a lack of HDI values, while Greenland was not analyzed because it is not part of the UNDP and Global Footprint Network databases.



2.7 Prioritizations and Trade-offs of Nature's Contributions to People

Policy-making regarding nature and its contributions to people requires approaches that allow decision-makers to maximize their time, resources and effectiveness. Two such tools are prioritization and trade-off analyses. We lay out two specific aspects to consider: i) prioritization by experts regarding specific NCP required to meet development targets and ii) trade-offs that take into account value and stakeholder plurality.

To determine the relative importance of specific NCP for attaining the SDG, the Americas Regional Assessment conducted a Delphi evaluation-consensus process among its network of experts. The purpose of this exercise was to provide guidance regarding the most important NCP that a decision-maker would need to incorporate into policies to attain these SDGs. All experts involved in the Assessment (Chairs, Coordinating Lead Authors, Lead Authors and Fellows) were invited to participate in this evaluation, which consisted of an expert assessment of the "top three" NCP for each SDG. As per the Delphi methodology (see Landeta, 2006), this process consisted of four steps: 1) An initial survey was conducted based on each individual's determination between 0 and 3 NCP a policy-maker would need to prioritize most to achieve each SDG; 2) Survey coordinators then synthesized the results into summary tables and figures; 3) This synthesis information then was provided only to those experts who responded to the survey in step 1, and they were asked to compare the group's collective knowledge to their individual responses. Upon reflection, each respondent was offered the opportunity to a) modify their NCP/SDG evaluation in a second survey or b) keep their responses as originally submitted; 4) Finally, the product of the entire process was developed from the final answers based on step 3 responses. This iterative process, inherent to the Delphi methodology, is meant to facilitate group learning and to achieve a greater level of consensus and precision in the establishment of this prioritization.

Results are presented as the percentage of respondents who identified each NCP as a "top 3" for a given SDG. These values were color-coded to indicate the level of consensus among respondents, with darker red indicating a greater level of agreement between experts (Figure 2.37, see also Figure 10 of the Summary for Policy Makers (SPM)). In addition, we calculated the importance value (IV, maximum = 2) of each NCP, considering its "relative abundance" among all responses identified as a priority for any of the SDGs (n_i / $n_{total,}$ where n_i is the number of times a specific NCP was identified as priority across all 17 SDGs and n_{total} = 31 responses x 17 SDGs = 547) and "relative frequency" (f_i / f_{total}, where f_i is the number of SDGs that included a specific NCP as priority, and f_{total} is all 17 SDGs).

We found that some NCP/SDG relationships are intuitive and have a clear consensus among respondents. For example, Food and Feed is essential to overcoming SDG1 (End poverty) and SDG2 (Zero hunger). The two NCP related to water regulation also are clearly necessary for SDG6 (Clean water and santiation). Other SDGs, though, lack a clear relationship with priority NCP. For example, SDGs 8, 9, 10 and 11 fall into this category, and overall we can say that their relationship with nature is either less direct or more multi-faceted, given their focus on such aspects of development as promoting sustained and inclusive development based on resilient infrastructure.

If we consider the categories of material, non-material and regulating NCP, material NCP are related to SDGs 1, 2, 3, 7 and 12, which can be explained by their importance to food security, energy security and health, as well as reducing the ecological footprint by changing patterns of production and consumption. Non-material NCP were related to SDGs 4, 5, 16 and 17, given their importance for subjective aspects of development, including education and gender equity, but also because they take into account intangible values related to global policies and cooperation to achieve sustainability. Finally, the regulating NCP are specific to climate, water and soil and are related to SDGs 6, 13, 14 and 15; these show some most consistent declines across the units of analysis (Figure SPM 10).

Figure 2.37 Priority nature's contributions to people (NCP) for achieving the Sustainable Development Goals (SDGs).

To determine the NCP that policy-makers could prioritize to achieve specific SDGs, the Americas Assessment conducted a Delphi process to elicit expert opinions from its authors and to establish levels of consensus regarding the three most important NCP for each SDG. Blank cells indicate no responses, and the intensity of the color red within cells illustrates the level of consensus among experts (% of respondents who prioritized a NCP for a specific SDG). Source: Data collected by C.B. Anderson, C. Simao Seixas & O. Barbosa and figure prepared by J. Diaz in R software package.

CATEGORIES OF NATURE'S CONTRIBUTIONS TO PEOPLE (NCP) OF IPBES	SUSTAINABLE DEVELOPMENT GOALS																
	SDG1	SDG2	SDG3	SDG4	SDG5	SDG6	SDG7	SDG8	SDG9	SDG10	SDG11	SDG12	SDG13	SDG14	SDG15	SDG16	SDG17
Food and Feed	30	28	14	5	4	0	0	11	0	18	3	23	2	8	7	6	3
Materials and assistance	21	4	2	1	5	1	11	11	16	2	7	8	0	0	3	1	3
Energy	16	5	1	4	2	3	31	12	9	9	7	13	7	0	1	2	4
Medicinal, biochemical and genetic resources	2	3	22	0	2	0	1	0	2	3	2	2	0	0	3	0	0
Learning and inspiration	2	3	5	28	19	0	3	9	13	8	3	9	0	2	1	21	21
Supporting identities	0	1	7	21	24	0	0	12	2	14	4	6	0	0	2	22	18
Physical and psychological experiences	0	0	17	13	10	0	0	0	2	1	4	0	0	0	0	4	3
Maintenance of options	6	1	1	10	7	1	9	15	11	15	4	4	4	2	7	17	13
Climate Regulation	3	2	2	0	0	14	11	3	2	4	7	0	28	8	5	1	1
Regulation of freshwater quantity, flow and timing	2	9	3	0	0	30	4	1	1	4	5	3	9	5	5	1	0
Regulation of freshwater and coastal water quality	1	2	3	0	0	27	0	0	2	1	3	2	1	26	0	0	1
Regulation of hazards and extreme events	1	1	2	0	0	8	6	2	11	1	17	0	15	4	2	0	1
Habitat creation and maintenance	2	1	0	1	0	0	4	2	4	1	10	3	10	6	23	1	0
Regulation of air quality	0	0	1	0	0	0	0	0	0	0	9	1	4	0	1	0	0
Regulation of organisms detrimental to humans	2	0	10	2	0	3	0	0	0	1	2	0	1	0	0	0	0
Pollination and dispersal of seeds and other propagules	2	17	0	0	0	0	0	0	0	0	0	3	0	0	11	0	0
Regulation of ocean acidification	0	1	0	0	0	1	1	1	1	1	0	1	5	23	0	0	0
Formation, protection and decontamination of soils and sediments	2	15	0	1	0	3	0	2	1	0	0	7	0	5	15	0	1

0 1—5 6—10 11—15 16—20 21—25 26—31

The importance value, which integrates both the number of times a NCP was prioritized by experts and the frequency of SDGs for which it was prioritized, demonstrates that material, non-material and regulating NCP must be considered to achieve the SDGs. In the case of Maintenance of Options, which obtained the greatest importance value (Table 2.25), it is transversal to SDGs and is also a transversal to all three NCP categories. These IV scores do not, however, suggest the importance of a specific NCP to a given SDG. Rather they demonstrate the overall importance to the suite of SDGs. For example, Pollination and Seed Dispersal is ranked low overall, but are crucial to SDG2 (Zero hunger) (see Table 2.25).

Table 2.25. Importance values (IV) were calculated for each of nature's contributions to people regarding its role in the achievement of all Sustainable Development Goals. Maximum IV = 2.

NCP	Category	IV
Maintenance of options	---	1.24
Energy	M	1.18
Learning and inspiration	N	1.16
Food and feed	M	1.13
Materials and assistance	M	1.06
Climate regulation	R	1.00
Regulation of freshwater quantity, flow and timing	R	0.98
Supporting identities	N	0.96
Regulation of hazards and extreme events	R	0.90
Habitat creation and maintenance	R	0.89
Regulation of freshwater and coastal water quality	R	0.78
Formation, protection, decontamination of soils & sediments	R	0.69
Medicinal, biochemical and genetic resources	M	0.67
Regulation of ocean acidification	R	0.60
Physical and psychological experiences	N	0.57
Regulation of organisms detrimental to humans	R	0.45
Regulation of air quality	R	0.32
Pollination and dispersal of seeds and other propagules	R	0.30

Such trade-offs between NCP are inherent in decision-making. Indeed, trade-offs occur when an ecosystem loses or has reduced one or more NCP to increase or gain another NCP, for instance, through economic development or restoration of some previous more natural state (e.g. Elmqvist et al., 2010). The analysis of these trade-offs examines not only the overall gain in human quality of life, but how those benefits are distributed; that is, who loses and who wins. A frequent objective of government trade-off analysis is to seek the mix of "co-benefits" that maximizes human well-being over both present and future generations (see example in Box 2.6).

Trade-off analysis is an essential aspect of any decision to invest in the protection or restoration of NCP (Leader-Williams et al., 2010). Figure 2.38 (Foley et al., 2005) illustrates the types of trade-offs that may occur when natural landscapes are protected or converted to agricultural use, or when some of the ecosystem services of agricultural landscapes are restored. This agricultural example applies generally to any land or water development. For example, electing to maintain and protect natural landscapes foregoes the benefits of agricultural development but maintains a potentially wide variety of benefits from natural services (see Figure 2.38 left). On the other hand, electing to farm comes at the cost of these other NCP to obtain the benefits provided by intensive agriculture. However, the maximum human benefit may be produced when natural services are partially restored in farmed landscapes, while retaining sustainable crop production. Nonetheless, an adequate cost-benefit analysis of these trade-offs must take into account the fact that some benefits are valued in monetary units (e.g. agricultural commodities) while other are often not acceptably measured in economic terms (e.g. sustainable livelihoods, sense of place) (Fig. 2.37 right). Preserving biodiversity, also identified in Figure 2.38, denies destructive use and the monetary measurement of the non-use value that justifies the preservation is controversial and not accepted all stakeholders (NRC, 2005). Therefore, government agencies concerned with development policies that reconcile environmental and social outcomes are required to recognize the different units of measure for use and nonuse values and integrate this subjectivity into the benefits that are being traded off and analyzed. It has been clearly shown that in governance and decision making, inequities in the distribution of benefits among stakeholders must be considered in addition to the risks that management plans may fail (Hanley & Spash, 1993; Boardman et al., 2011), which requires taking into account this value plurality in the trade-off assessment.

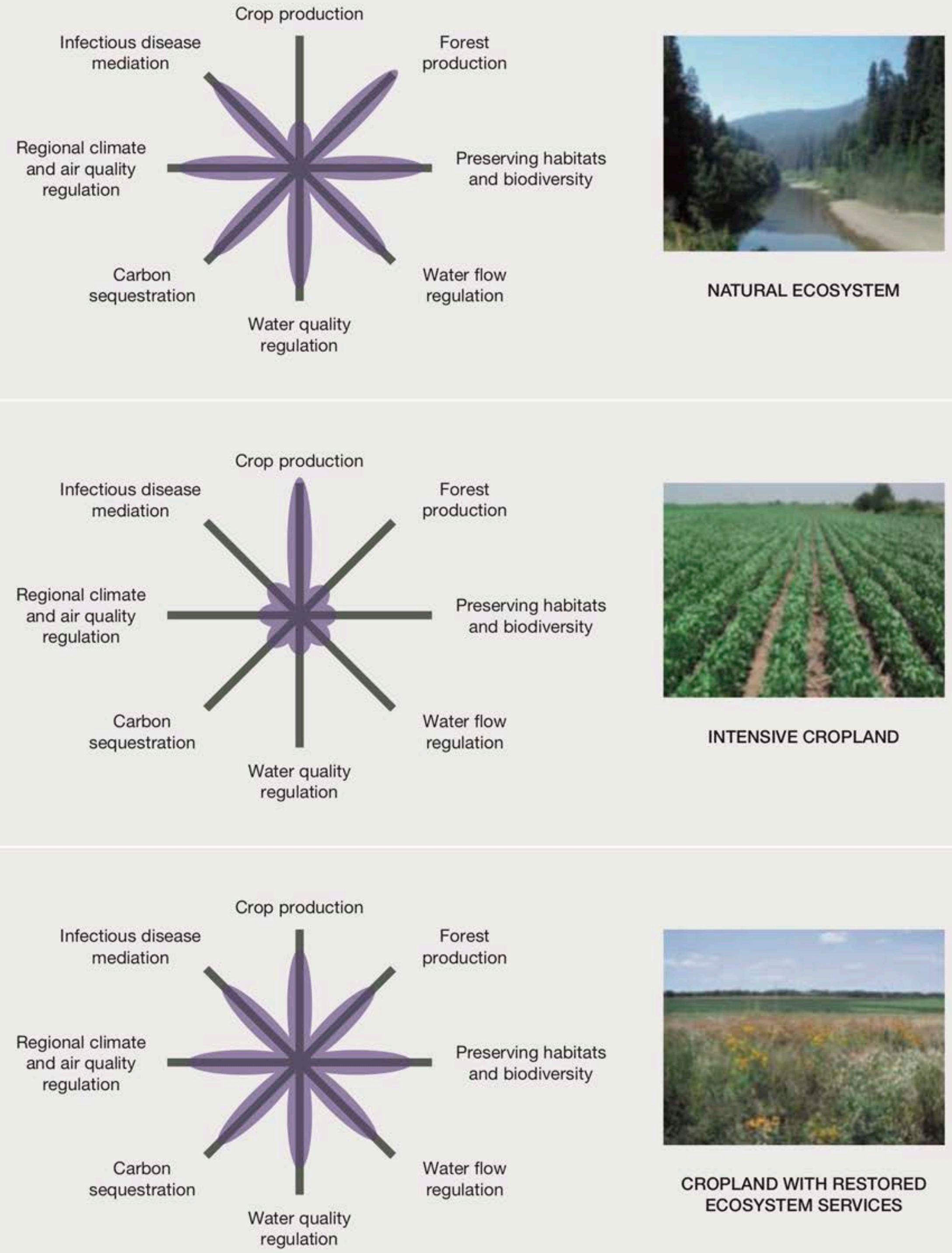
Trade-off analysis may be relatively simple when the benefits from different services and the costs of protecting, developing, or restoring them can be measured in the same units of value, such as monetary currency (Boardman et al., 2011). For instance, tourism is a major resource for such mountain economies, and studies in many regions have shown that protection of watersheds provides greater economic value than resource extraction (The Mountain Institute, 1998; UNEP, 2008). In Figure 2.38, most ecosystem services are material or regulating NCP and have use value, which can, in general, be measured monetarily (NRC, 2005; Tietenberg & Lewis, 2014; Harris & Roach, 2014). Nevertheless, we may also need to address non-use values, such as trade-offs between use and protection to preserve options for future generations, between material and non-material NCP, or when the same NCP are valued through different values systems, or even when they are provided at different scales. An example of the latter is given by Raudsepp-Hearne et al. (2010), who observed in Quebec, Canada, that landscape-scale trade-offs between provisioning (material NCP) and almost all regulating and cultural (regulating and non-material NCP) ecosystem services, and they show that a greater diversity of ecosystem services is positively correlated with the provision of regulating ecosystem services. Trade-off analysis is also complicated by consideration of who benefits and who bears the costs, and by inexperience in communicating across different value systems and worldviews.

Below, we pose some of the questions that we tried to address in the prior sections (with different levels of success, in part due to knowledge gaps).

- What are the trade-offs of expanding cropland and rangeland over natural ecosystems to feed animals or other nations?
- What are the trade-off between food security and energy security regarding biomass production?
- What are the trade-offs of building hydropower plants and dams (such as Belo Monte in Brazilian Amazon) over the land of indigenous groups with high risk of culture and language extinction, and the loss of aquatic and terrestrial biodiversity?
- What are the trade-offs of mining over indigenous land or protected areas (e.g. the development of oil sands extraction and pipelines built over First Nations and Métis settlements living in northern Alberta, Canada)?
- What are the trade-offs of implementing no-take protected areas for conserving future options while creating "conservation refugees" and decimating cultures? No-take protected areas (International Union for Conservation of Nature, category I and II) are an important strategy to maintain options for the future, but may affect cultural continuity and livelihoods of displaced indigenous groups and local communities?
- What are the trade-off of GMO production, conventional agriculture and organic production?
- What are the trade-offs of fisheries closures for conservation purpose and increased aquaculture production (including all its environmental impacts) versus the devastation of thousands of local fisher livelihoods?
- What are the trade-offs of increased urbanization and economic growth versus health and livelihood security?
- What are the trade-offs of water usage for agriculture production versus human needs, and the needs of resident species?
- What are the trade-offs of conserving watersheds versus extracting its resources?
- What are the consequences of protecting or restoring landscapes for food, water, raw materials, energy, and cultural security?
- How much could the health, pleasure, and other aspects of well-being for future generations be compromised by a massive loss to extinction of options maintained by species?
- Is service restoration always an option, or are the risks and uncertainties often too great to rely on as a correction for what turns out to be a bad development decision?
- How can the uncertainty associated with subjective comparisons of relative service value, as expressed in different units of measure, be improved?

These and many others are questions that decision–makers face when planning policies, strategies, actions. The consequences of trade-offs made during decision making may extend well into the future and require complex trend analysis for forecasting future needs across the full spectrum of benefits and costs. This chapter is intended to shed some light on these questions by showing how NCP affect quality of life in different biomes and subregions of the Americas.

Figure 2 38 **Conceptual framework for comparing land use and trade-offs in ecosystem services. Source: Foley *et al.* (2015).**



2.8 Knowledge gaps

Despite many advances in ecosystem service science and the connection of ecosystems to human well-being, more comprehensive assessments of costs, benefits and values are necessary to more fully understand the relationship of nature and quality of life at the regional and subregional scales. There is still a narrow focus on one or few services (NCP), and without a proper understanding of their relationships and interactions (Bennett et al., 2009). More holistic evaluations should put greater attention on the role of regulating and non-material (cultural) NCP when assessing land change processes and well change in the ocean. Admittedly, there are more difficulties in quantifying and valuing these less tangible NCP, which are more amenable to the standardization of monetary values via market mechanisms (sections 2.2.5, 2.2.6, 2.2.7). At the same time, we should point out that non-material NCP, like identities, are closely linked to human rights considerations, which in fact makes economic, cost-benefit type analyses inappropriate, and in violation of international agreements. Plus, we observed frequent gaps in databases, due to the fact that most social data is collected at the political scale, while ecological information is often specific to an ecosystem or biome. Even so, some political entities (e.g. Greenland) are almost entirely absent from global databases managed by the UN, World Bank and others, thus limiting country-level comparisons on all aspects of both social and ecological data.

2.9 References

Aburto-Oropeza, O., Erisman, B., Galland, G. R., Mascareñas-Osorio, I., Sala, E., & Ezcurra, E. (2011). Large recovery of fish biomass in a no-take marine reserve. *PLoS ONE*, *6*(8). https://doi.org/10.1371/journal.pone.0023601

Adams, W. M., & Hutton, J. (2007). People, Parks and Poverty: Political Ecology and Biodiversity Conservation. *Conservation and Society*, *5*(2), (pp. 147–183).

Afrane, Y. A., Zhou, G., Lawson, B. W., Githeko, A. K., & Yan, G. (2006). Effects of microclimatic changes caused by deforestation on the survivorship and reproductive fitness of Anopheles gambiae in western Kenya highlands. *American Journal of Tropical Medicine and Hygiene*, *74*(5), (pp. 772–778). https://doi.org/74/5/772 {pii}

Agard, J., & Cropper, A. (2007). Caribbean Sea Ecosystem Assessment. A sub-global component of the Millenium Ecosystem Assessment. *Caribbean Marine Studies Special Ed*.

Agardy, T., di Sciara, G. N., & Christie, P. (2011). Mind the gap: Addressing the shortcomings of marine protected areas through large scale marine spatial planning. *Marine Policy*, *35*(2), (pp. 226–232). https://doi.org/https://doi.org/10.1016/j.marpol.2010.10.006

Aizen, M. A., Vázquez, D. P., & Smith-Ramírez, C. (2002). Historia natural y conservación de los mutualismos planta-animal del bosque templado de Sudamerica austral. *Revista Chilena de Historia Natural*, *75*(1), (pp. 79–97). https://doi.org/10.4067/S0716-078X2002000100008

Akimoto, H. (2003). *Global Air Quality and Pollution*. *Science*, *302*(5651), (pp. 1716–1719). https://doi.org/10.1126/science.1092666

Alcamo, J., Florke, M., & Marker, M. (2007). Future long-term changes in global water resources driven by socio-economic and climatic changes. *Hydrological Sciences Journal*, *52*(2), (pp. 247–275). https://doi.org/10.1623/hysj.52.2.247

Ali, A., & Abdulai, A. (2010). The Adoption of Genetically Modified Cotton and Poverty Reduction in Pakistan. *Journal of Agricultural Economics*, *61*(1), (pp. 175–192). https://doi.org/10.1111/j.1477-9552.2009.00227.x

Ali, H., Khan, E., & Sajad, M. A. (2013). Phytoremediation of heavy metals—Concepts and applications. *Chemosphere*, *91*(7), (pp. 869–881). https://doi.org/https://doi.org/10.1016/j.chemosphere.2013.01.075

Alkemade, R., Reid, R. S., van den Berg, M., de Leeuw, J., & Jeuken, M. (2013). Assessing the impacts of livestock production on biodiversity in rangeland ecosystems. *Proceedings of the National Academy of Sciences*, *110*(52), (pp. 20900–20905). https://doi.org/10.1073/pnas.1011013108

Allan, B. F., Langerhans, R. B., Ryberg, W. A., Landesman, W. J., Griffin, N. W., Katz, R. S., Oberle, B. J., Schutzenhofer, M. R., Smyth, K. N., de St. Maurice, A., Clark, L., Crooks, K. R., Hernandez, D. E., McLean, R. G., Ostfeld, R. S., & Chase, J. M. (2009). Ecological correlates of risk and incidence of West Nile virus in the United States. *Oecologia*, *158*(4), 699–708. https://doi.org/10.1007/s00442-008-1169-9

Allan, B. F., Tallis, H., Chaplin-Kramer, R., Huckett, S., Kowal, V. A., Musengezi, J., Okanga, S., Ostfeld, R. S., Schieltz, J., Warui, C. M., Wood, S. A., & Keesing, F. (2017). Can integrating wildlife and livestock enhance ecosystem services in central Kenya? *Frontiers in Ecology and the Environment*, *15*(6), 328–335. https://doi.org/10.1002/fee.1501

Allem, A. C., Mendes, R. A., Salomão, A. N., & Burle, M. L. (2001). The primary gene pool of cassava (*Manihot esculenta* Crantz subspecies *esculenta*, Euphorbiaceae). *Euphytica*, *120*, (pp. 127–132).

Allen, C. D., & Breshears, D. D. (1998). Drought-induced shift of a forest-woodland ecotone: Rapid landscape response to climate variation. *Proceedings of the National Academy of Sciences*, *95*(25), (pp. 14839–14842). https://doi.org/10.1073/pnas.95.25.14839

Alonso-Castro, A., Juárez-Vázquez, M., & Campo-Xolalpa, N. (2016). Medicinal Plants from Mexico , Central America , and the Caribbean Used as Immunostimulants. *Evidence-Based Complementary and AlternativeMedicine*, *2016*, (pp. 1–16). https://doi.org/10.1155/2016/4017676

Alston, J. M., Chan-Kang, C., M.C.Marra, Pardey, P. G., & Wyatt., T. J. (2000). *A meta analysis of rates of return to agricultural R&D: ex pede herculem?* Washington, D.C.

Altrichter, M. (2006). *Interacciones entre la gente y la fauna en el Chaco Argentino*. Córdoba, Argentina:

Secertaría de Ambiente y Desarrollo Sustentable.

Álvarez, J., & Shany, N. (2012). Una experiencia de gestión participativa de la biodiversidad con comunidades amazónicas. *Revista Peruana de Biología*, *19*(2). https://doi.org/10.15381/rpb.v19i2.846

Alves, R. R., & Alves, H. N. (2011). The faunal drugstore: Animal-based remedies used in traditional medicines in Latin America. *Journal of Ethnobiology and Ethnomedicine*, *7*(1), (pp. 9). https://doi.org/10.1186/1746-4269-7-9

Ammour, T., Windevoxhel, N., & Sención, G. (2000). Economic valuation of mangrove ecosystems and subtropical forests in Central America. In M. Dore & R. Guevera (Eds.), *Sustainable forest management and global climate change: selected case studies from the Americas* (pp. 166–197). Cheltenham, UK: Edward Elgar Publishing Ltd.

Amundson, R., & Jenny, H. (1997). On a State Factor Model of Ecosystems. *BioScience*, *47*(8), (pp. 536–543). https://doi.org/10.2307/1313122

Andam, K. S., Ferraro, P. J., Pfaff, A., Sanchez-Azofeifa, G. A., & Robalino, J. A. (2008). Measuring the effectiveness of protected area networks in reducing deforestation. *Proceedings of the National Academy of Sciences*, *105*(42), (pp. 16089–16094). https://doi.org/10.1073/pnas.0800437105

Anderson, I., Robson, B., Connolly, M., Al-Yaman, F., Bjertness, E., King, A., Tynan, M., Madden, R., Bang, A., Coimbra, C. E. A., Pesantes, M. A., Amigo, H., Andronov, S., Armien, B., Obando, D. A., Axelsson, P., Bhatti, Z. S., Bhutta, Z. A., Bjerregaard, P., Bjertness, M. B., Briceno-Leon, R., Broderstad, A. R., Bustos, P., Chongsuvivatwong, V., Chu, J., Deji, Gouda, J., Harikumar, R., Htay, T. T., Htet, A. S., Izugbara, C., Kamaka, M., King, M., Kodavanti, M. R., Lara, M., Laxmaiah, A., Lema, C., Taborda, A. M. L., Liabsuetrakul, T., Lobanov, A., Melhus, M., Meshram, I., Miranda, J. J., Mu, T. T., Nagalla, B., Nimmathota, A., Popov, A. I., Poveda, A. M. P., Ram, F., Reich, H., Santos, R. V, Sein, A. A., Shekhar, C., Sherpa, L. Y., Skold, P., Tano, S., Tanywe, A., Ugwu, C., Ugwu, F., Vapattanawong, P., Wan, X., Welch, J. R., Yang, G., Yang, Z., & Yap, L. (2016). Indigenous and tribal peoples' health (The Lancet–Lowitja Institute Global Collaboration): a population study. *The Lancet*, *388*(10040), 131–157. https://doi.org/10.1016/S0140-6736(16)00345-7

Antunes, A. P., Fewster, R. M., Venticinque, E. M., Peres, C. A., Levi, T., Röhe, F., & Shepard Jr., G. H. (2016). Empty forest or empty rivers? A century of commercial hunting in Amazonia. *Science Advances*, *2*(10), e1600936–e1600936. https://doi.org/10.1126/sciadv.1600936

Araújo, C. A. C., Waniek, P. J., & Jansen, A. M. (2009). An Overview of Chagas Disease and the Role of Triatomines on Its Distribution in Brazil. *Vector-Borne and Zoonotic Diseases*, *9*(3), (pp. 227–234). https://doi.org/10.1089/vbz.2008.0185

Aronson, J., Clewell, A. F., Blignaut, J. N., & Milton, S. J. (2006). Ecological restoration: A new frontier for nature conservation and economics. *Journal for Nature Conservation*, *14*(3–4), (pp. 135–139). https://doi.org/10.1016/j.jnc.2006.05.005

Aronson, M. F. J., La Sorte, F. A., Nilon, C. H., Katti, M., Goddard, M. A., Lepczyk, C. A., Warren, P. S., Williams, N. S. G., Cilliers, S., Clarkson, B., Dobbs, C., Dolan, R., Hedblom, M., Klotz, S., Kooijmans, J. L., Kühn, I., Macgregor-Fors, I., McDonnell, M., Mörtberg, U., Pysek, P., Siebert, S., Sushinsky, J., Werner, P., & Winter, M. (2014). A global analysis of the impacts of urbanization on bird and plant diversity reveals key anthropogenic drivers. *Proceedings. Biological Sciences*, *281*(1780), 20133330. https://doi.org/10.1098/rspb.2013.3330

Arriagada, R. A., Sills, E. O., Ferraro, P. J., & Pattanayak, S. K. (2015). Do payments pay off? Evidence from participation in Costa Rica's PES program. *PLoS ONE*, *10*(7), (pp. 1–17). https://doi.org/10.1371/journal.pone.0131544

Ashworth, L., Quesada, M., Casas, A., Aguilar, R., & Oyama, K. (2009). Pollinator-dependent food production in Mexico. *Biological Conservation*, *142*(5), (pp. 1050–1057). https://doi.org/10.1016/j.biocon.2009.01.016

Association of Zoos and Aquariums (AZA). (2017). *Zoo and aquarium statistics*. Retrieved May 2, 2017, from https://www.aza.org/zoo-and-aquarium-statistics

Avnery, S., Mauzerall, D. L., Liu, J., & Horowitz, L. W. (2011). Global crop yield reductions due to surface ozone exposure: 2. Year 2030 potential crop production losses and economic damage under two scenarios of O3 pollution. *Atmospheric Environment*, *45*(13), (pp. 2297–2309).

https://doi.org/10.1016/j.atmosenv.2011.01.002

Ayala, J. (1997). *Utilización de la fauna silvestre del grupo étnico Ayoreode en la comunidad Tobité, Santa Cruz, Bolivia*. UAGRM.

Ayala, R., Gonzalez, H. V., & Engel, S. M. (2013). Mexican Stingless Bees (Hymenoptera:Apidae): Diversity, Distribution, and Indigenous Knowledge. In *Pot-Honey: A Legacy of Stingless Bees* (pp. 1–654). https://doi.org/10.1007/978-1-4614-4960-7

Baeza, A., Santos-Vega, M., Dobson, A. P., & Pascual, M. (2017). The rise and fall of malaria under land-use change in frontier regions. *Nature Ecology & Evolution*, *1*(5), (pp. 108). https://doi.org/10.1038/s41559-017-0108

Bahia, F. N. C., Seixas, S. C., Araujo, L. G., Farinaci, J. S., & Chamy, P. (2013). Implementation of a National Park over traditional lands of the Trindade community in Paraty, Brazil. In T. C. Magro, L. M. Rodrigues, D. F. Silva Filho, J. L. Polizel, & J. Leahy (Eds.), *Protected Areas and Place Making Conference* (pp. 46–51). Foz do Iguaçú, PR: Forestry Sciences Departament – ESALQ/USP.

Balee, W. (1985). *The Kalapalo Indians of central Brazil. Holt, Rinehart and Winston. Human Ecology* (Vol. 13).

Balée, W. (2013). *Cultural Forests of the Amazon: A Historical Ecology of People and Their Landscapes.* University of Alabama Press.

Balée, W. L. (1993). *Footprints of the Forest Ka'apor Ethnobotany—the Historical Ecology of Plant Utilization by an Amazonian People*. New York, NY, USA: Columbia University Press.

Balvanera, P., Castillo, A., & Martínez-Harms, M. J. (2011). Ecosystem Services in Seasonally Dry Tropical Forests. In R. Dirzo, H. S. Young, H. A. Mooney, & G. Ceballos (Eds.), *Seasonally Dry Tropical Forests* (pp. 259–277). Island Press/Center for Resource Economics.

Balvanera, P., Castillo, A., & Martínez-Harms, M. J. (2011). Ecosystem Services in Seasonally Dry Tropical Forests. In R. Dirzo, H. S. Young, H. A. Mooney, & G. Ceballos (Eds.), *Seasonally Dry Tropical Forests* (pp. 259–277). Island Press/Center for Resource Economics.

Barbier, E. B., & Enchelmeyer, B. S. (2014). Valuing the storm surge protection service of US Gulf Coast wetlands. *Journal of Environmental Economics and Policy*, *3*(2), (pp. 167–185). https://doi.org/10.1080/21606544.2013.876370

Barbier, E. B., & Strand, I. (1998). Valuing Mangrove-Fishery Linkages -- A Case Study of Campeche, Mexico. *Environmental and Resource Economics*, *12*(2), (pp. 151–166). https://doi.org/10.1023/A:1008248003520

Barbosa, O., & Villagra, P. (2015). *Socio-Ecological Studies in Urban and Rural Ecosystems in Chile*. (R. Rozzi & et al., Eds.), *Earth Stewardship Ecology and Ethics*. Chile: Springer International Publishing Switzerland. https://doi.org/10.1007/978-3-319-12133-8_19

Barbosa, O., Tratalos, J. A., Armsworth, P. R., Davies, R. G., Fuller, R. A., Johnson, P., & Gaston, K. J. (2007). Who benefits from access to green space? A case study from Sheffield, UK. *Landscape and Urban Planning*, *83*(2–3), (pp. 187–195). https://doi.org/10.1016/j.landurbplan.2007.04.004

Barboza, G. E., Cantero, J. J., Núñez, C., Pacciaroni, A., & Espinar, L. A. (2009). Medicinal plants: A general review and a phytochemical and ethnopharmacological screening of the native Argentine Flora. *Kurtziana*, *34*(1–2).

Bardach, A., Ciapponi, A., Rey-Ares, L., Rojas, J. I., Mazzoni, A., Glujovsky, D., Valanzasca, P., Romano, M., Elorriaga, N., Dantur Juri, M. J., & Boulos, M. (2015). Epidemiology of Malaria in Latin America and the Caribbean from 1990 to 2009: Systematic Review and Meta-Analysis. *Value in Health Regional Issues*, *8*, 69–79. https://doi.org/10.1016/j.vhri.2015.05.002

Barnes, B. V., Zak, D. R., Denton, S. R., & Spurr, S. H. (1998). *Forest Ecology, 4th Edition*. New York: John Wiley & Sons, Inc.

Barrett, C. B. (2010). Measuring Food Insecurity. *Science*, *327*(February), (pp. 825–828). https://doi.org/10.1126/science.1182768

Barrios, Y., Ramírez, N., Ramírez, E., Sánchez, E., & Del Castillo, R. (2010). Importancia de los polinizadores en la reproducción de seis especies de subpáramo del Pico Naiguata (Parque Nacional el Ávila-Venezuela). *Acta Botánica Venezuélica*, *33*(2), 213–231.

Basso, E. B. (1973). *The Kalapalo Indians of central Brazil*. *Case studies in cultural anthropology*.

Bastos, G. C., & Petrere, M. (2010). Small-scale marine fisheries in the municipal district of Florianópolis, Santa Catarina, Brazil. *Brazilian Journal of Biology*, *70*(4), (pp. 947–953). https://doi.org/10.1590/S1519-69842010000500005

Baumgardner, D., Varela, S., Escobedo, F. J., Chacalo, A., & Ochoa, C. (2012). The role of a peri-urban forest on air quality improvement in the Mexico City megalopolis. *Environmental Pollution*, *163*(April), (pp. 174–183). https://doi.org/10.1016/j.envpol.2011.12.016

Begossi, A. (1998). Cultural and ecological resilience among caiçaras of the Atlantic Forest and caboclos of the Amazon, Brazil. In F. Berkes & C. Folke (Eds.), *Linking Social and Ecological Systems* (pp. 129–157). Cambridge University Press.

Benavidez, G. (1999). *Gestión local de recursos naturales y agroecología. Un proceso de recuperación de la base productive de recursos naturales en el municipio Comanche*. *RURALTER*, *18*, (pp. 183–189).

Béné, C., Macfadayen, G., & Allison, E. H. (2007). *Increasing the contribution of small-scale fisheries to poverty alleviation and food security* (Fisheries and AquacultureTechnical Papers No. 481). Rome, Italy.

Benedict, M. A., & McMahon, E. T. (2002). *Green infrastructure: Linking landscapes and communities*. Washington, D.C.: Island Press.

Bennett, E. M., Peterson, G. D., & Gordon, L. J. (2009). Understanding relationships among multiple ecosystem services. *Ecology Letters*, *12*(12), (pp. 1394–1404). https://doi.org/10.1111/j.1461-0248.2009.01387.x

Berbés-Blázquez, M., González, J. A., & Pascual, U. (2016). Towards an ecosystem services approach that addresses social power relations. *Current Opinion in Environmental Sustainability*, *19*, (pp. 134–143). https://doi.org/10.1016/j.cosust.2016.02.003

Bergstrom, J. C., Stoll, J. R., Titre, J. P., & Wright, V. L. (1990). Economic value of wetlands-based recreation. *Ecological Economics*, *2*(2), (pp. 129–147). https://doi.org/http://dx.doi.org/10.1016/0921-8009(90)90004-E

Berkes, F. (2008). *Sacred Ecology.* Routledge.

Berman, M. G., Jonides, J., & Kaplan, S. (2008). The Cognitive Benefits of Interacting With Nature. *Psychological Science*, *19*(12), (pp. 1207–1212). https://doi.org/10.1111/j.1467-9280.2008.02225.x

Bervoets, T. (2010). *Working Paper on the Economic Valuation of Country St . Maarten ' s Coral Reef Resources*. St. Maarten; Netherlands Antilles.

Birch, J. C., Newton, A. C., Aquino, C. A., Cantarello, E., Echeverria, C., Kitzberger, T., Schiappacasse, I., & Garavito, N. T. (2010). Cost-effectiveness of dryland forest restoration evaluated by spatial analysis of ecosystem services. In *Proceedings of the National Academy of Sciences* (Vol. 107, pp. 21925–21930). https://doi.org/10.1073/pnas.1003369107

Boardman, A. E., Greenberg, D. H., R.Vining, A., & Weimer, D. L. (2011). *Cost-Benefit Analysis (4th Edition)*. Saddle River, NJ: Prentice Hall College Div.

Bonito, L. T., Hamdoun, A., & Sandin, S. A. (2016). Evaluation of the global impacts of mitigation on persistent, bioaccumulative and toxic pollutants in marine fish. *PeerJ*, *4*, e1573. https://doi.org/10.7717/peerj.1573

Bormann, F. H., & Likens, G. E. (1967). Nutrient Cycling. *Science*, *155*(3761), 424 LP-429. Retrieved from http://science.sciencemag.org/content/155/3761/424.abstract

Borré, K. (1991). *Seal Blood, Inuit Blood, and Diet: A Biocultural Model of Physiology and Cultural Identity. Medical Anthropology Quarterly*, *5*(1), (pp. 48–62). https://doi.org/10.1525/maq.1991.5.1.02a00080

Bosch, J. M., & Hewlett, J. D. (1982). A review of catchment experiments to determine the effect of vegetation changes on water yield and evapotranspiration. *Journal of Hydrology*, *55*(1–4), (pp. 3–23). https://doi.org/10.1016/0022-1694(82)90117-2

Bossio, D., Geheb, K., & Critchley, W. (2010). Managing water by managing land: Addressing land degradation to improve water productivity and rural livelihoods. *Agricultural Water Management*, *97*(4), (pp. 536–542). https://doi.org/10.1016/j.agwat.2008.12.001

Botkin, D. B., Saxe, H., Araújo, M. B., Betts, R., Bradshaw, R. H. W., Cedhagen, T., Chesson, P., Dawson, T. P., Etterson, J. R., Faith, D. P., Ferrier, S., Guisan, A., Hansen, A. S., Hilbert, D. W., Loehle, C., Margules, C., New, M., Sobel, M. J., & Stockwell, D. R. B. (2007). Forecasting the Effects of Global Warming on Biodiversity. *BioScience*, *57*(3), 227–236. https://doi.org/10.1641/B570306

Bowes, M. D., & Krutilla, J. V. (1989). *Multiple-use Management: The Economics of Public Forest Lands*. Routledge. https://doi.org/doi:10.4324/9781315060576

Boyles, J. G., Cryan, P. M., McCracken, G. F., & Kunz, T. H. (2011). Economic Importance of Bats in Agriculture. *Science*, *332*(6025), 41 LP-42. Retrieved from http://science.sciencemag.org/content/332/6025/41.abstract

Bradley, S. R., Vuille, M., Diaz, H. F., & Vergara, W. (2006). Threats to Water Supplies in the Tropical Andes. *Science*, *312*(June), (pp. 1755–1756).

Brauman, K., Daily, G., Duarte, T., & Mooney, H. (2007). The Nature and Value of Ecosystem Services: An Overview Highlighting Hydrologic Services. *Annual Review of Environment and Resources*. https://doi.org/10.1146/annurev.energy.32.031306.102758

Bravo de Guenni, L., Cardoso, M., Goldammer, J., Hurtt, G., & Mata, L. J. (2009). *Regulation of Natural Hazards : Floods and Fires*. *Millennium Ecosystem Assessment: Current State & Trends Assessment*. Retrieved from http://www.millenniumassessment.org/en/Condition.aspx

Bravo-Monroy, L., Tzanopoulos, J., & Potts, S. G. (2015). Ecological and social drivers of coffee pollination in Santander, Colombia. *Agriculture, Ecosystems & Environment*, *211*, (pp. 145–154). https://doi.org/10.1016/j.agee.2015.06.007

Breaux, A., Farber, S., & Day, J. (1995). Using Natural Coastal Wetlands Systems for Wastewater Treatment: An Economic Benefit Analysis. *Journal of Environmental Management*, *44*(3), (pp. 285–291). https://doi.org/http://dx.doi.org/10.1006/jema.1995.0046

Brooks, K. N., Ffolliott, P. F., & Agner, J. A. (2012). *Hydrology and the Management of Watersheds* (Fourth Edi). Wiley-Blackwell.

Brooks, T. M., Akçakaya, H. R., Burgess, N. D., Butchart, S. H. M., Hilton-Taylor, C., Hoffmann, M., Juffe-Bignoli, D., Kingston, N., MacSharry, B., Parr, M., Perianin, L., Regan, E. C., Rodrigues, A. S. L., Rondinini, C., Shennan-Farpon, Y., & Young, B. E. (2016a). Analysing biodiversity and conservation knowledge products to support regional environmental assessments. *Scientific Data*, *3*, 160007. https://doi.org/10.1038/sdata.2016.7

Brooks, T. M., Akçakaya, H. R., Burgess, N. D., Butchart, S. H. M., Hilton-Taylor, C., Hoffmann, M., Juffe-Bignoli D., Kingston N., MacSharry B., Parr M., Perianin L., Regan E. C., Rodrigues A. S. L., Rondinini C., Shennan-Farpon Y., Young, B. E. (2016b). Data from: Analysing biodiversity and conservation knowledge products to support regional environmental assessments. *Dryad Digital Repository.* https://doi.org/doi:10.5061/dryad.6gb90.2

Brosi, B. J., Armsworth, P. R., & Daily, G. C. (2008). Optimal design of agricultural landscapes for pollination services. *Conservation Letters*, *1*(1), (pp. 27–36). https://doi.org/10.1111/j.1755-263X.2008.00004.x

Brown, A. V., Brown, K. B., Jackson, D. C., & Pierson, W. K. (2005). Lower Mississippi River and its tributaries. In A. C. Benke & C. E. Cushing (Eds.), *Rivers of North America* (pp. 231–281). New York, NY: Elsevier Academic Press.

Brown, T. C., & Froemke, P. (2012). Improved Measures of Atmospheric Deposition Have a Negligible Effect on Multivariate Measures of Risk of Water-Quality Impairment: Response from Brown and Froemke. *BioScience*, *62*(7), (pp. 621–622). https://doi.org/10.1525/bio.2012.62.7.19

Brundtland & World Commission on Environment. (1987). *Report of the World Commission on Environment and Development: Our Common Future*. Oxford University Press.

Bruner, A. G., Gullison, R. E., Rice, R. E., & da Fonseca, G. A. B. (2001). Effectiveness of Parks in Protecting Tropical Biodiversity. *Science*, *291*(5501), 125 LP-128. Retrieved from http://science.sciencemag.org/content/291/5501/125.abstract

Bryan, B. A. (2013). *Incentives, land use, and ecosystem services: Synthesizing complex linkages*. *Environmental Science and Policy*, *27*, 124–134. https://doi.org/10.1016/j.envsci.2012.12.010

Bryant, D., Burke, L., McManus, J., & Spalding, M. (1998). *Reefs at Risk – A Map Based Indicator of Threats to the World's Coral Reefs*. https://doi.org/10.1016/0022-0981(79)90136-9

Burakowski, E., & Magnusson, M. (2012). *Climate Impacts on the Winter Tourism Economy in the United States*. *National Resources Defense Council*.

Burgett, M., Daberkow, S., Rucker, R., & Thurman, W. (2010). U.S. pollination markets: recent changes and historical perspectives. *American Bee Journal*, *150*(1), (pp. 35–41).

Burke, L., & Maidens, J. (2004). Reefs at Risk in the Caribbean. Washington, D.C.: *World Resources Institute (WRI).* Retrieved from http://pdf.wri.org/reefs_caribbean_front.pdf

Burke, L., Reytar, K., Spalding, M., & Perry, A. (2011). Reefs at risk revisited. National Geographic. Washington, D.C.: *World Resources Institute (WRI).* https://doi.org/10.1016/0022-0981(79)90136-9

Buytaert, W., & Breuer, L. (2013). Water resources in South America: sources and supply, pollutants and perspectives. In *Proceedings of the IAHS-IAPSO-IASPEI Assembly* (Vol. 359, pp. 1–9). Gouthenburg, Sweden.

Buytaert, W., & De Bièvre, B. (2012). Water for cities: The impact of climate change and demographic growth in the tropical Andes. *Water Resources Research*, *48*(8), n/a-n/a. https://doi.org/10.1029/2011WR011755

Buytaert, W., Célleri, R., De Bièvre, B., Cisneros, F., Wyseure, G., Deckers, J. & Hofstede, R. (2006). Human impact on the hydrology of the Andean páramos. *Earth-Science Reviews*, *79*(1–2), (pp. 53–72). https://doi.org/10.1016/j.earscirev.2006.06.002

Caballero, J., Casas, A., Cortés, L., & Mapes, C. (1998). Patrones en el conocimiento, uso y manejo de plantas en pueblos indigenas de México. *Estudios Atacameños*, (16), (pp. 181–195). Retrieved from http://www.jstor.org/stable/25674716

Cabrera, M., Seijo, J., Euan, J., & Pérez, E. (1998). Economic values of ecological services from a mangrove ecosystem. *International Newsletter of Coastal Management*. Retrieved from http://pdf.usaid.gov/pdf_docs/PNACP233.pdf

Cáceres, D. M. (2015). Accumulation by Dispossession and Socio-Environmental Conflicts Caused by the Expansion of Agribusiness in Argentina. *Journal of Agrarian Change*, *15*(1), (pp. 116–147). https://doi.org/10.1111/joac.12057

Callicott, J. B. (1989). *In Defense of the Land Ethic: Essays in Environmental Philosophy*. Albany, NY: New York University Press.

Camacho-Valdez, V., Ruiz-Luna, A., Ghermandi, A., & Nunes, P. A. L. D. (2013). Valuation of ecosystem services provided by coastal wetlands in northwest Mexico. *Ocean & Coastal Management*, *78*, (pp. 1–11). https://doi.org/10.1016/j.ocecoaman.2013.02.017

Caminade, C., Kovats, S., Rocklov, J., Tompkins, A. M., Morse, A. P., Colón-González, F. J., Stenlund, H., Martens, P., & Lloyd, S. J. (2014). Impact of climate change on global malaria distribution. *Proceedings of the National Academy of Sciences of the United States of America*, *111*(9), 3286–3291. https://doi.org/10.1073/pnas.1302089111

Carlyle-Moses, D. E., & Gash, J. H. C. (2011). Rainfall Interception Loss by Forest Canopies BT - Forest Hydrology and Biogeochemistry: Synthesis of Past Research and Future Directions. In D. F. Levia, D. Carlyle-Moses, & T. Tanaka (Eds.), *Forest Hydrology and Biogeochemistry* (pp. 407–423). Dordrecht: Springer Netherlands. https://doi.org/10.1007/978-94-007-1363-5_20

Carneiro da Cunha, M., & Morim de Lima, A. G. (2017). Amazonian indigenous peoples and biodiversity. In B. Baptiste, D. Pacheco, M. Carneiro da Cunha, & S. Diaz (Eds.), *Knowing our Lands and Resources: Indigenous and Local Knowledge of Biodiversity and Ecosystem Services in the Americas* (pp. 63–81). Paris: UNESCO.

Carpenter, K. A., & Halbritter, R. (2001). Beyond the Ethnic Umbrella and the Buffalo: Some Thoughts on American Indian Tribes and Gaming. *Gaming Law Review*, *5*(4), (pp. 311–327). https://doi.org/10.1089/109218801750430290

CARSEA. (2007). *Caribbean Sea Ecosystem Assessment (CARSEA), A sub-global component of the Millennium Ecosystem Assessment (MA)*. (J. Agard, A. Cropper, & K. García, Eds.), *Caribbean Marine Studies* (Vol. Special Ed). Retrieved from file:///E:/Documents and Settings/Administrador/Escritorio/Articulo de profesora/Referencias del articulo/Agard y Cropper.2007.pdf

Carvalho, I. S. H. de. (2014). *A "pecuária geraizeira" e a conservação da biodiversidade no cerrado do Norte de Minas*. *Sustentabilidade Em Debate*, *5*(3), (pp. 19–36).

Castellanos, A., Chaparro-Narváez, P., Morales-Plaza, C. D., Alzate, A., Padilla, J., Arévalo, M., & Herrera, S. (2016). Malaria in gold-mining areas in Colombia. *Memórias Do Instituto Oswaldo Cruz*, *111*(1), (pp. 59–66). https://doi.org/10.1590/0074-02760150382

Castillo, A., Magaña, A., Pujadas, A., Martínez, L., & Godínez, C. (2005). Understanding the Interaction of Rural People with Ecosystems: A Case Study in a Tropical Dry Forest of Mexico. *Ecosystems*, *8*(6), (pp. 630–643). https://doi.org/10.1007/s10021-005-0127-1

Cech, T. V. (2010). *Principles of Water Resources: History, Development, Management, and Policy* (3rd Editio). Hoboken, NJ, USA: John Wiley & Sons, Inc.

CEPAL/FAO/IICA. (2012). *Perspectivas de la agricultura y del desarrollo rural en las Américas: Una mirada hacia América Latina y el Caribe*. Santiago de Chile.

Chacón, M. I. (2009). *Darwin y la domesticación de plantas en las Américas: El caso del maíz y frijol. Acta Biológica Colombiana*, *14*, (pp. 351–364).

Chambers, R., & Conway, R. G. (1991). Sustainable rural livelihoods: practical concepts for the 21st century (No. 296). *IDS Discussion Paper* (Vol. 296). https://doi.org/ISBN 0 903715 58 9

Chan, K. M. A., Balvanera, P., Benessaiah, K., Chapman, M., Díaz, S., Gómez-Baggethun, E., Gould, R., Hannahs, N., Jax, K., Klain, S., Luck, G. W., Martín-López, B., Muraca, B., Norton, B., Ott, K., Pascual, U., Satterfield, T., Tadaki, M., Taggart, J., & Turner, N. (2016). Why protect nature? Rethinking values and the environment. *PNAS*, *113*(6), (pp. 1462–1465). https://doi.org/doi/10.1073/pnas.1525002113

Chan, K. M. A., Guerry, A. D., Balvanera, P., Klain, S., Satterfield, T., Basurto, X., Bostrom, A., Chuenpagdee, R., Gould, R., Halpern, B. S., Hannahs, N., Levine, J., Norton, B., Ruckelshaus, M., Russell, R., Tam, J., & Woodside, U. (2012). Where are Cultural and Social in Ecosystem Services? A Framework for Constructive Engagement. *BioScience*, *62*(8), (pp. 744–756). https://doi.org/10.1525/bio.2012.62.8.7

Chapin III, F. S., Matson, P. A., & Mooney, H. A. (2011). *Principles of Terrestrial Ecosystem Ecology* (Second Edi). New York: Springer.

Charles, A., Westlund, L., Bartley, D. M., Fletcher, W. J., Garcia, S., Govan, H., & Sanders, J. (2016). Fishing livelihoods as key to marine protected areas: insights from the World Parks Congress. *Aquatic Conservation: Marine and Freshwater Ecosystems, 26*, 165–184. https://doi.org/10.1002/aqc.2648.

Chevallier, P., Pouyaud, B., Suarez, W., & Condom, T. (2011). Climate change threats to environment in the tropical Andes: glaciers and water resources. *Regional Environmental Change*, *11*(S1), (pp. 179–187). https://doi.org/10.1007/s10113-010-0177-6

Chhabra, A., Geist, H. J., Houghton, R. A., Haberl, H., Braimoh, A., Vlek, P. L., Patz, J. A., Xu, J., Ramankutty, N., Coomes, O. T., & Lambin, E. F. (2006). Multiple Impacts of Land-Use/Cover Change. In E. F. Lambin & H. J. Geist (Eds.), *Land-use and Land-Cover Change: Local Processes and Global Impacts* (pp. 41–70). Berlin: Springer-Verlag.

Chmura, G. L., Anisfeld, S. C., Cahoon, D. R., & Lynch, J. C. (2003). Global carbon sequestration in tidal, saline wetland soils. *Global Biogeochemical Cycles*, *17*(4), n/a-n/a. https://doi.org/10.1029/2002GB001917

Choque Quispe, M. E. (2017). *La crianza de la llama y la gestión de los conocimientos tradicionalessobre la diversidad biológica y los ecosistemasen Corque Marka, departamento de Oruro, Bolivia*. In B. Baptiste, D. Pacheco, M. C. da Cunha, & S. Diaz (Eds.), *Knowing our Lands and Resources: Indigenous and Local Knowledge of Biodiversity and Ecosystem Services in the Americas*. Paris: UNESCO.

Claro, R. M., Santos, M. A. S., Oliveira, T. P., Pereira, C. A., Szwarcwald, C. L., Malta, D. C., Claro, R. M., Santos, M. A. S., Oliveira, T. P., Pereira, C. A., Szwarcwald, C. L., & Malta, D. C. (2015). Unhealthy food consumption related to chronic non- communicable diseases in Brazil : National Health Survey, 2013. *Epidemiologia e Serviços de Saúde*, *24*(2), 257–265. https://doi.org/10.5123/S1679-49742015000200008

Clay, J. (2004). *World Agriculture and the Environment: A Commodity-By-Commodity Guide To Impacts And Practices (p. 570).* Island Press.

Clayton, S., & Myers, G. (2009). *Conservation Psychology: Understanding and Promoting Human Care for Nature*. Wiley-Blackwell.

Clement, C. R., Denevan, W. M., Heckenberger, M. J., Junqueira, A. B., Neves, E. G., Teixeira, W. G., &

Woods, W. I. (2015). The domestication of Amazonia before European conquest. *Proceedings. Biological Sciences / The Royal Society*, *282* (1812), 20150813. https://doi.org/10.1098/rspb.2015.0813

Cohen, J. E. (1995). *How Many People Can the Earth Support?* New York: Norton. https://doi.org/10.2307/2137520

Coleman-Jensen, A., Rabbitt, M. P., Gregory, C. A., & Singh, A. (2016). *Household Food Security in the United States in 2015-A report summary from the Economic Research Service*. Washington, D.C. Retrieved from https://www.ers.usda.gov/webdocs/publications/err215/err-215.pdf

Collard, I. F., & Foley, R. A. (2002). Latitudinal patterns and environmental determinants of recent human cultural diversity: do humans follow biogeographical rules? *Evolutionary Ecology Research*, *4*, (pp. 371–383).

Collins, R., & Rutherford, K. (2004). Modelling bacterial water quality in streams draining pastoral land. *Water Research*, *38*(3), (pp. 700–712). https://doi.org/10.1016/j.watres.2003.10.045

Coloma, C., Hoffman, J. S., & Crosby, A. (2006). Suicide Among Guaraní Kaiowá and Nandeva Youth in Mato Grosso do Sul, Brazil. *Archives of Suicide Research*, *10*(2), (pp. 191–207). https://doi.org/10.1080/13811110600662505

Committee on the Status of Pollinators. (2007). *Status of Pollinators in north america*. Washington, D.C.: The National Academies Press.

Constantino, P. de A. L. (2016). *Deforestation and hunting effects on wildlife across amazonian indigenous lands. Ecology and Society*, *21*(2). https://doi.org/10.5751/ES-08323-210203

Cooley, S. R., & Doney, S. C. (2009). Anticipating ocean acidification’s economic consequences for commercial fisheries. *Environmental Research Letters*, *4*(2), 24007. https://doi.org/10.1088/1748-9326/4/2/024007

Copa, M. E., & Townsend, W. R. (2004). Aprovechamiento de la fauna por dos comunidades Tsimane’: un subsidio del bosque a la economía familiar. *Revista Boliviana de Ecología Y Conservación Ambiental*, (16), 41–48. Retrieved from http://biblat.unam.mx/es/revista/revista-boliviana-de-ecologia-y-conservacion-ambiental/articulo/aprovechamiento-de-la-fauna-por-dos-comunidades-tsimane-un-subsidio-del-bosque-a-la-economia-familiar

Cormier, L., & Urbani, B. (2008). The ethnoprimatiology of spider monkeys (Atlele spp.) from past to present. In C. Campbel (Ed.), *Spider Monkeys: Biology, Ecology and Evolution of the genus Ateles*. Cambridge University Press.

Cortés, F. I. A., Pérez, M. Ló., & MogollÓn, H. M. (2012). Mexico’s Water Challenges for the 21st Century. In *Water Resources in Mexico* (pp. 21–38). Springer Berlin Heidelberg. https://doi.org/10.1007/978-3-642-05432-7_2

Corvalan, C., Hales, S., & McMichael, A. (2005). *Ecosystems and Human Well-being: Health Synthesis*. Geneva Switzerland.

Costa, M. H., & Foley, J. A. (2000). Combined Effects of Deforestation and Doubled Atmospheric CO 2 Concentrations on the Climate of Amazonia. *Journal of Climate*, *13*(1), (pp. 18–34). https://doi.org/10.1175/1520-0442(2000)013<0018:CEODAD>2.0.CO;2

Costanza, R., & Farber, S. (1987). The economic value of wetland systems. *Journal of Environmental Management*, *24*, (pp. 41–51).

Costanza, R., D’Arge, R., de Groot, R., Farber, S., Grasso, M., Hannon, B., Limburg, K., Naeem, S., O’Neill, R. V., Paruelo, J., Raskin, R. G., Sutton, P., & van den Belt, M. (1997). The value of the world’s ecosystem services and natural capital. *Nature*, *387*(6630), 253–260. https://doi.org/10.1038/387253a0

Costanza, R., de Groot, R., Sutton, P., van der Ploeg, S., Anderson, S. J., Kubiszewski, I., Farber, S., & Turner, R. K. (2014). Changes in the global value of ecosystem services. *Global Environmental Change*, *26*, 152–158. https://doi.org/10.1016/j.gloenvcha.2014.04.002

Costanza, R., Pérez-Maqueo, O., Martinez, M. L., Sutton, P., Anderson, S. J., & Mulder, K. (2008). The Value of Coastal Wetlands for Hurricane Protection. *AMBIO: A Journal of the Human Environment*, *37*(4), (pp. 241–248). https://doi.org/10.1579/0044-7447(2008)37{241:TVOCWF}2.0.CO;2

Costanza, R., Wilson, M., Troy, A., Voinov, A., Liu, S., & D’Agostino, J. (2006). *The value of New Jersey’s ecosystem services and natural capital*. *… and The New Jersey …*. Trenton, NJ. Retrieved from

http://training.fws.gov/EC/Resources/nrdar/injury_quantification/NJvaluationpart2.pdf
Costello, A., Abbas, M., Allen, A., Ball, S., Bell, S., Bellamy, R., Friel, S., Groce, N., Johnson, A., Kett, M., Lee, M., Levy, C., Maslin, M., McCoy, D., McGuire, B., Montgomery, H., Napier, D., Pagel, C., Patel, J., de Oliveira, J. A. P., Redclift, N., Rees, H., Rogger, D., Scott, J., Stephenson, J., Twigg, J., Wolff, J., & Patterson, C. (2009). Managing the health effects of climate change. *The Lancet*, *373*(9676), 1693–1733. https://doi.org/10.1016/S0140-6736(09)60935-1
Cruz-Coutiño, A. (2014). *Cacao Soconusco Apuntes sobre Chiapas, México y Centromaérica*. UNICACH : El Aguaje.
CTO. (2015). *Caribbean Tourism Orgnization: Latest Statistics 2015*. *Caribben Tourism Organization*. Retrieved from file:///C:/Users/julio/Desktop/SOD LITERATURE/Lattab15_FINAL.pdf
Cuellar, R. (1997). *Aprovechamiento de la fauna Silvestre en una comunidad de agricultores: los guanraní de Akae, Santa Cruz*. UMSA.
Cunniff, S., & Schwartz, A. (2015). *Performance of Natural Infrastructure and Nature-based Measures as Coastal Risk Reduction Features*. Washington, D.C.
da Xavier, S. C. C., Roque, A. L. R., dos Lima, V. S., Monteiro, K. J. L., Otaviano, J. C. R., da Silva, L. F. C. F., & Jansen, A. M. (2012). Lower richness of small wild mammal species and chagas disease risk. *PLoS Neglected Tropical Diseases*, *6*(5). https://doi.org/10.1371/journal.pntd.0001647
Daily, G. C. (1997). *Nature's Services: Society Dependence on Natural Ecosystems*. Island Press.
Dallimer, M., Tinch, D., Hanley, N., Irvine, K. N., Rouquette, J. R., Warren, P. H., Maltby, L., Gaston, K. J., & Armsworth, P. R. (2014). Quantifying Preferences for the Natural World Using Monetary and Nonmonetary Assessments of Value. *Conservation Biology*, *28*(2), 404–413. https://doi.org/10.1111/cobi.12215
Danver, S. L. (2015). *Native Peoples of the World: An Encyclopedia of Groups, Cultures and Contemporary Issues* (Vol. 4). Routledge. Retrieved from https://books.google.com/books?id=vf4TBwAAQBAJ&pgis=1
Darvill, R., & Lindo, Z. (2016). The inclusion of stakeholders and cultural ecosystem services in land management trade-off decisions using an ecosystem services approach. *Landscape Ecology*, *31*(3), (pp. 533–545). https://doi.org/10.1007/s10980-015-0260-y
Dawson, T. P., Jackson, S. T., House, J. I., Prentice, I. C., & Mace, G. M. (2011). Beyond predictions: biodiversity conservation in a changing climate. *Science*, *332*(6025), (pp. 53–58). https://doi.org/10.1126/science.1200303
De Beenhouwer, M., Aerts, R., & Honnay, O. (2013). A global meta-analysis of the biodiversity and ecosystem service benefits of coffee and cacao agroforestry. *Agriculture, Ecosystems & Environment*, *175*(Supplement C), (pp. 1–7). https://doi.org/https://doi.org/10.1016/j.agee.2013.05.003
de Castro, E. V, Souza, T. B., & Thapa, B. (2015). Determinants of Tourism Attractiveness in the National Parks of Brazil. *Parks*, *21*(2), (pp. 51–62). https://doi.org/10.2305/IUCN.CH.2014.PARKS-21-2EVDC.en
de Padua, L. S. Bunyapraphatsara, N. Lemmens, R. H. M. J. (1999). Medicinal and poisonous plants. Vol. 1. Leiden, Netherlands: *Plants Resources of South-East Asia 12/1*.
De Silva, T. (1997). *Industrial utilization of medicinal plants in developing countries*. (FAO, Ed.), *Medicinal Plants for Forest Conservation and Health care*. Rome, Italy.
DeBarry, P. A. (2004). *Watersheds: Processes, Assessment and Management*. Hoboken, NJ, USA: John Wiley & Sons.
Delach, A. (2012). *Harnessing Nature: The Ecosystem Approach to Climate-Change Preparedness*.
Denevan, W. (1966). *The Aboriginal Cultural Geography of the Llanos de Mojos of Bolivia*. *Ibero-Americana No. 48* (Vol. 48). https://doi.org/10.2307/2511415
Desmarchelier, C. (2010). Neotropics and natural ingredients for pharmaceuticals: why isn't South American biodiversity on the crest of the wave? *Phytotherapy Research*, n/a-n/a. https://doi.org/10.1002/ptr.3114
Detwiler, R. P. (1986). *Land use change and the global carbon cycle: the role of tropical soils. Biogeochemistry, 2(1)*, (pp. 67–93). https://doi.org/10.1007/BF02186966
Di Stasi, L. C., Oliveira, G. P., Carvalhaes, M. A., Queiroz-Junior, M., Tien, O. S., Kakinami, S. H., & Reis, M. S.

(2002). Medicinal plants popularly used in the Brazilian Tropical Atlantic Forest. *Fitoterapia*, *73*(1), (pp. 69–91). https://doi.org/10.1016/S0367-326X(01)00362-8

Dias, J. E., & Laureano, L. C. (2010). *Farmacopéia popular do Cerrado*. Retrieved from http://medcontent.metapress.com/index/A65RM03P4874243N.pdf

Diaz, R. J., & Rosenberg, R. (2008). Spreading dead zones and consequences for marine ecosystems. *Science*, *321*(5891), (pp. 926–929). https://doi.org/10.1126/science.1156401

Díaz, S., Demissew, S., Joly, C., Lonsdale, W. M., & Larigauderie, A. (2015). A Rosetta Stone for Nature ' s Benefits to People. *Plos Biology*, (pp. 1–8). https://doi.org/10.1371/journal.pbio.1002040

Dimitri, C., & Greene, C. (2002). Recent Growth Patterns in the U.S. Organic Foods Market. *Agriculture Information Bulletin* (Vol. 777). https://doi.org/10.1111/j.1440-1819.2009.02038.x

Dise, N. B., Ashmore, M.R., Belyazid, S., Bobbink, R., De Vries, W., Erisman, J.W., Spranger, T.**,** Stevens, C. & van den Berg, L. (2011). Nitrogen as a threat to European Biodiversity. In M. A. Sutton, C. M. Howard, J. W. Erisman, G. Billen, A. Bleeker, P. Grennfelt, H.V. Grinsven & B. Grizzetti (Eds.), *The European Nitrogen Assessment*. Cambridge, UK: Cambridge University Press.

Donoso, P. J., Frêne, C., Flores, M., Moorman, M. C., Oyarzún, C. E., & Zavaleta, J. C. (2014). Balancing water supply and old-growth forest conservation in the lowlands of south-central Chile through adaptive co-management. *Landscape Ecology*, *29*(2), (pp. 245–260). https://doi.org/10.1007/s10980-013-9969-7

dos Santos, G. M., & Antonini, Y. (2008). The traditional knowledge on stingless bees (Apidae: Meliponina) used by the Enawene-Nawe tribe in western Brazil. *Journal of Ethnobiology and Ethnomedicine*, *4*, (pp. 19). https://doi.org/10.1186/1746-4269-4-19

Dowsley, M. (2010). *The Value of a Polar Bear: Evaluating the Role of a Multiple-Use Resource in the Nunavut Mixed Economy*. *Arctic Anthropology* , *47*(1), (pp. 39–56). https://doi.org/10.1353/arc.0.0035

Dudley, N. (2008). *Guidelines for Applying Protected Area Management Categories*. (N. Dudley, Ed.). Gland, Switzerland: IUCN. Retrieved from https://books.google.co.uk/books?id=pq4oEg58_08C

Dudley, N., Belokurov, A., Higgins-Zogib, L., Hockings, M., Stolton, S., & Burgess, N. (2007). *Tracking progress in managing protected areas around the world*. Gland, Switzerland.

Dudley, N., S. Stolton, A. Belokurov, L. Krueger, N. Lopoukhine, K. MacKinnon, T. Sandwith and N. Sekhran [editors] (2009); Natural Solutions: Protected areas helping people cope with climate change, IUCNWCPA, TNC, UNDP, WCS, The World Bank and WWF, Gland, Switzerland, Washington DC and New York, USA.

Dunne, T., & Leopold, L. B. (1978). *Water in Environmental Planning*. W H Freeman & Co.

Ehrlich, P. R., & Mooney, H. A. (1983). Extinction, Substitution, and Ecosystem Services. *BioScience*, *33*(4), (pp. 248–254). https://doi.org/10.2307/1309037

EIA. (2016). *International Energy Statistics*. Retrieved from https://www.eia.gov/beta/international/data/browser/#/?vs=INTL.44-1-AFRC-QBTU.A&vo=0&v=H&start=1980&end=2014

Elias, M. A. S., Borges, F. J. A., Bergamini, L. L., Franceschinelli, E. V., & Sujii, E. R. (2017). Climate change threatens pollination services in tomato crops in Brazil. *Agriculture, Ecosystems & Environment*, *239*, (pp. 257–264). https://doi.org/10.1016/j.agee.2017.01.026

Elmqvist, T., Maltby, E., Barker, T., Mortimer, M., & Perrings, C. (2010). Biodiversity, ecosystems and ecosystem services. In P. Kumar (Ed.), *The Economics of Ecosystems and Biodiversity: Ecological and Economic Foundations* (pp. 41–112). New York, NY: Routledge.

Emperaire, L. (2017). Saberes tradicionais e diversidade das plantas cultivadas na Amazônia. In B. Baptiste, D. Pacheco, M. Carneiro da Cunha, & S. Diaz (Eds.), *Knowing our Lands and Resources: Indigenous and Local Knowledge of Biodiversity and Ecosystem Services in the Americas* (pp. 41–62). Paris: UNESCO.

Erdoes, R., & Ortiz, A. (1985). *American Indian Myths and Legends*. New York: Pantheon Books.

Erickson, C. (2006). El valor actual de los Camellones de cultivo precolombinos: Experiencias del Perú y Bolivia. In F. Valdez (Ed.), *Agricultura ancestral. Camellones y albarradas: Contexto social, usos y retos del pasado y del presente* (pp. 315–339). Quito, Ecuador: Ediciones Abya-Yala.

Erickson, C. L. (2010). The Transformation of Environment into Landscape: The Historical Ecology of

Monumental Earthwork Construction in the Bolivian Amazon. *Diversity*, *2*(4), (pp. 618–652). https://doi.org/10.3390/d2040619

Escobedo, F. J., Wagner, J. E., Nowak, D. J., De la Maza, C. L., Rodriguez, M., & Crane, D. E. (2008). Analyzing the cost effectiveness of Santiago, Chile's policy of using urban forests to improve air quality. *Journal of Environmental Management*, *86*(1), (pp. 148–157). https://doi.org/10.1016/j.jenvman.2006.11.029

Espinosa, C. (2005). *Payment for Water-Based Environmental Services: Ecuador's Experiences, Lessons Learned and Ways Forward* (Nature and Economics No. Technical Paper No. 2). Colombo.

Faber, S., & Male, T. (2012). *Plowed Under: How Crop Subsidies Contribute to Massive Habitat Losses*.

FAO, IFAD, & WFP. (2015). *The State of Food Insecurity in the World: Meeting the 2015 international hunger targets: taking stock of uneven progress. FAO, IFAD and WFP*. https://doi.org/I4646E/1/05.15

FAO. (2016). *AQUASTAT Main Database (2016 Edition)*. Retrieved from http://www.fao.org/nr/water/aquastat/data/query/index.html?lang=en

Farley, J., & Costanza, R. (2010). Payments for ecosystem services: From local to global. *Ecological Economics*, *69*(11), (pp. 2060–2068). https://doi.org/https://doi.org/10.1016/j.ecolecon.2010.06.010

Farnsworth, N. R., & Soejarto, D. D. (1991). Global Importance of Medicinal Plants. In H. Synge, O. Akerele, & V. Heywood (Eds.), *Conservation of Medicinal Plants* (pp. 25–52). Cambridge: Cambridge University Press. https://doi.org/DOI: 10.1017/CBO9780511753312.005

Fauset, S., Johnson, M. O., Gloor, M., Baker, T. R., Monteagudo M., A., Brienen, R. J. W., Feldpausch, T. R., Lopez-Gonzalez, G., Malhi, Y., ter Steege, H., Pitman, N. C. A., Baraloto, C., Engel, J., Pétronelli, P., Andrade, A., Camargo, J. L. C., Laurance, S. G. W., Laurance, W. F., Chave, J., Allie, E., Vargas, P. N., Terborgh, J. W., Ruokolainen, K., Silveira, M., Aymard C., G. A., Arroyo, L., Bonal, D., Ramirez-Angulo, H., Araujo-Murakami, A., Neill, D., Hérault, B., Dourdain, A., Torres-Lezama, A., Marimon, B. S., Salomão, R. P., Comiskey, J. A., Réjou-Méchain, M., Toledo, M., Licona, J. C., Alarcón, A., Prieto, A., Rudas, A., van der Meer, P. J., Killeen, T. J., Marimon Junior, B.-H., Poorter, L., Boot, R. G. A., Stergios, B., Torre, E. V., Costa, F. R. C., Levis, C., Schietti, J., Souza, P., Groot, N., Arets, E., Moscoso, V. C., Castro, W., Coronado, E. N. H., Peña-Claros, M., Stahl, C., Barroso, J., Talbot, J., Vieira, I. C. G., van der Heijden, G., Thomas, R., Vos, V. A., Almeida, E. C., Davila, E. Á., Aragão, L. E. O. C., Erwin, T. L., Morandi, P. S., de Oliveira, E. A., Valadão, M. B. X., Zagt, R. J., van der Hout, P., Loayza, P. A., Pipoly, J. J., Wang, O., Alexiades, M., Cerón, C. E., Huamantupa-Chuquimaco, I., Di Fiore, A., Peacock, J., Camacho, N. C. P., Umetsu, R. K., de Camargo, P. B., Burnham, R. J., Herrera, R., Quesada, C. A., Stropp, J., Vieira, S. A., Steininger, M., Rodríguez, C. R., Restrepo, Z., Muelbert, A. E., Lewis, S. L., Pickavance, G. C., & Phillips, O. L. (2015). Hyperdominance in Amazonian forest carbon cycling. *Nature Communications*, *6*(1), 6857. https://doi.org/10.1038/ncomms7857

Federal Provincial and Territorial Governments of Canada. (2012). *2012 Canadian Nature Survey: Awareness, participation, and expenditures in nature-based recreation, conservation, and subsistence activities*. Ottawa, ON.

Fekete, B. M., Vorosmarty, C. J., & Grabs, W. (1999). *Global, composite runoff fields based on observed river discharge and simulated water balances*.

Feldman, D. (2012). *Water*. Cambridge, UK: Polity Press.

Fennessy, S., & Craft, C. (2011). Agricultural conservation practices increase wetland ecosystem services in the Glaciated Interior Plains. *Ecological Applications*, *21*(3 SUPPL.), (pp. 49–64). https://doi.org/10.1890/09-0269.1

Fernandes, A., Medeiros, C., Bernardo, G., Ebone, M., Di Pietro, P., Assis, M., & Vasconcelos, F. (2012). Benefits and risks of fish consumption for the human health. *Revista de Nutrição*, *25*(2), (pp. 283–295). https://doi.org/10.1590/S1415-52732012000200010

Fernandes-Pinto, E. (2017). *Sítios Naturais Sagrados do Brasil: inspirações para o reencantamento das áreas protegidas*. Universidade Federal do Rio de Janeiro.

Ferraro, P. J., & Hanauer, M. M. (2011). Protecting Ecosystems and Alleviating Poverty with Parks and Reserves: "Win-Win" or Tradeoffs? *Environmental and Resource Economics*, *48*(2), (pp. 269–286). https://doi.org/10.1007/s10640-010-9408-z

Festa-Bianchet, M., & Côté, S. D. (2008). *Mountain Goats: Ecology, Behavior, and Conservation of an Alpine Ungulate*. Washington: Island Press.

Fialkowski, M. K., Okoror, T. A., & Boushey, C. J. (2012). The Relevancy of Community-Based Methods: Using Diet within Native American and Alaska Native Adult Populations as an Example. *Clinical and Translational Science*, *5*(3), (pp. 295–300). https://doi.org/10.1111/j.1752-8062.2011.00364.x

Finlay, J., Franke, T., McKay, H., & Sims-Gould, J. (2015). Therapeutic landscapes and wellbeing in later life: Impacts of blue and green spaces for older adults. *Health & Place*, *34*, (pp. 97–106). https://doi.org/https://doi.org/10.1016/j.healthplace.2015.05.001

Fischer, G., Hizsnyik, E., Prieler, S., Shah, M., & Van Velthuizen, H. (2009). *Biofuels and food security*. *OFID and IIASA report* (Vol. 1). https://doi.org/10.1111/j.1467-9353.2008.00425.x

Fischer, G., Velthuizen, H. Van, Shah, M., & Nachtergaele, F. (2002). Global Agro-ecological Assessment for Agriculture in the 21st Century : Methodology and Results (IIASA Rese). *Analysis* (Vol. RR-02-02). Laxenburg, Austria: IIASA/FAO. https://doi.org/http://www.iiasa.ac.at/Admin/PUB/Documents/RR-02-002.pdf

Fischer, T., Byerlee, D., Edmeades, G., Byerlee, D., Edmeades, G., & Fischer, T. (2014). Crop yields and global food security: will yield increase continue to feed the world*? ACIAR Monograph N° 158*. Camberra, Australia.

Fisher, B., Turner, R. K., & Morling, P. (2009). Defining and classifying ecosystem services for decision making. *Ecological Economics*, *68*(3), (pp. 643–653). https://doi.org/https://doi.org/10.1016/j.ecolecon.2008.09.014

Foley, J. A., Asner, G. P., Costa, M. H., Coe, M. T., DeFries, R., Gibbs, H. K., Howard, E. A., Olson, S., Patz, J., Ramankutty, N., & Snyder, P. (2007). Amazonia revealed: Forest degradation and loss of ecosystem goods and services in the Amazon Basin. *Frontiers in Ecology and the Environment*, *5*(1), 25–32. https://doi.org/10.1890/1540-9295(2007)5[25:ARFDAL]2.0.CO;2

Foley, J. a., Costa, M. H., Delire, C., Ramankutty, N., & Snyder, P. (2003). Green surprise? How terrestrial ecosystems could affect earth's climate. *Frontiers in Ecology and the Environment*, *1*(1), (pp. 38–44). https://doi.org/10.1890/1540-9295(2003)001{0038:GSHTEC}2.0.CO;2

Foley, J. A., Defries, R., Asner, G. P., Barford, C., Bonan, G., Carpenter, S. R., Chapin, F. S., Coe, M. T., Daily, G. C., Gibbs, H. K., Helkowski, J. H., Holloway, T., Howard, E. A., Kucharik, C. J., Monfreda, C., Patz, J. A., Prentice, I. C., Ramankutty, N., & Snyder, P. K. (2005). Global consequences of land use. *Science (New York, N.Y.)*, *309*(5734), 570–574. https://doi.org/10.1126/science.1111772

Fonnegra, R. G., & Jiménez, R. L. (2007). *Plantas Medicinales Aprobadas En Colombia*. Universidad de Antioquia.

Food and Agriculture Organisation (FAO). (2015a). *FAOSTAT Database*. Retrieved from http://faostat3.fao.org/home/E

Food and Agriculture Organisation (FAO). (2015b). *Global Forest Resources Assessment 2015: Desk Reference*. Rome. Retrieved from http://www.fao.org/3/a-i4808e.pdf

Food and Agriculture Organization (FAO). (1997). *Report of the World Food Summit*. Rome, Italy. Retrieved from http://www.fao.org/docrep/003/w3548e/w3548e00.htm#date01

Food and Agriculture Organization (FAO). (1999). *Background Paper 1: Agricultural Biodiversity. In FAO/Netherlands Conference on the Multifunctional Character of Agriculture and Land* (pp. 1–42). Retrieved from http://www.fao.org/ WAICENT/FAOINFO/SUSTDEV/EPdirect/EPdocs/ MFCAL.pdf

Food and Agriculture Organization (FAO). (2002). *World agriculture : towards 2015 / 2030. Summary report* (Vol. 20). Rome: FAO. https://doi.org/10.1016/S0264-8377(03)00047-4

Food and Agriculture Organization (FAO). (2006). *Food Security*. *Policy Brief*. Rome: FAO. https://doi.org/10.1016/j.jneb.2010.12.007

Food and Agriculture Organization (FAO). (2007). *Organic agriculture and access to food*. In M. Sligh & C. Christman (Eds.), *International Conference on Organic Agriculture and Food Security* (p. 27). Rome, Italy: FAO. Retrieved from http://scholar.google.com/scholar?hl=en&btnG=Search&q=intitle:Organic+Agriculture+and+Access+to+Food#0

Food and Agriculture Organization (FAO). (2009). *Pago por Servicios Ambientales en Áreas Protegidas en*

América Latina. *Programa FAO/OAPN: Fortalecimiento del Manejo Sostenible de los Recursos Naturales en las Áreas Protegidas de América Latina*. Retrieved from http://www.fao.org/3/a-i0822s.pdf

Food and Agriculture Organization (FAO). (2010). *The role of forests and forestry in the prevention and rehabilitation of lanslides in Asia*. *Forests ans landslides, The center for people and forests, FAO*.

Food and Agriculture Organization (FAO). (2011a). *The State of the World's Land and Water Resources for Food and Agriculture. Managing Systems at Risk*. New York: Earthscan. https://doi.org/10.4324/9780203142837

Food and Agriculture Organization (FAO). (2011b). *World Livestock 2011- Livestock in food security World*. (A. McLeod, Ed.), *World*. Rome, FAO.: FAO. https://doi.org/10.1080/00036841003742587

Food and Agriculture Organization (FAO). (2014a). *FOA Yearbook: Fishery and Aquaculture Statistic Summary Tables*. Rome, Italy: FAO.

Food and Agriculture Organization (FAO). (2014b). *The State of Food and Agriculture: Innovation in Family Farming*. Rome. https://doi.org/9789251073179

Food and Agriculture Organization (FAO). (2015). *FAO Statistical Pocketbook 2015*. *Food and Agriculture Organization of the United Nations*. Rome: FAO. https://doi.org/978-92-5-108802-9

Food and Agriculture Organization (FAO). (2016a). *2015 Global Forest Products Facts and Figures*. Rome: FAO. Retrieved from http://www.fao.org/3/a-i6669e.pdf

Food and Agriculture Organization (FAO). (2016b). *Global Forest Resources Assessment 2015*. *FAO Forestry* (Second Edi). Rome: FAO. https://doi.org/10.1002/2014GB005021

Food and Agriculture Organization (FAO). (2017). *FAOSTAT*. Retrieved August 25, 2017, from http://www.fao.org/faostat/en/#data

Food and Agriculture Organization of the United Nations Regional Office for the Latin America and the Caribbean (FAO RLC). (2014). *FAO STATISTICAL YEARBOOK 2014 Latin America and the Caribbean Food and Agriculture*. Santiago de Chile. Retrieved from http://www.fao.org/docrep/019/i3592e/i3592e.pdf

Fortescue, J. A. C. (1980). *Environmental Geochemistry: A Holistic Approach*. New York, NY: Springer-Verlag.

Francou, B., & Vincent, C. (2007). *Les glaciers à l'épreuve du climat*. Paris: IRD Editions et Editions.

Francou, B., Vuille, M., Favier, V., & Cáceres, B. (2004). New evidence for an ENSO impact on low-latitude glaciers: Antizana 15, Andes of Ecuador, 0°28'S. *Journal of Geophysical Research: Atmospheres*, *109*(D18), n/a-n/a. https://doi.org/10.1029/2003JD004484

Franzmann, A., Schwartz, C., & McCabe, R. E. (2007). *Ecology and Management of the North American Moose*. Boulder: University Press of Colorado.

Fredrickson, L. M., & Anderson, D. H. (1999). A QUALITATIVE EXPLORATION OF THE WILDERNESS EXPERIENCE AS A SOURCE OF SPIRITUAL INSPIRATION. *Journal of Environmental Psychology*, *19*(1), (pp. 21–39). https://doi.org/http://dx.doi.org/10.1006/jevp.1998.0110

Funk, J., Saunders, S., Sanford, T., Easely, T., & Markham, A. (2014). *Rocky Mountain Forests at Risk: Confronting Climate-driven Impacts from Insects, Wildfires, Heat, and Drought*. Cambridge, MA. Retrieved from www. ucsusa.org/forestsatrisk

FUSION. (2015). Factories get away with illegally dumping toxic chemicals into one of the country's main waterways, with potentially deadly consequences. Retrieved from http://interactive.fusion.net/river-of-death/

Gain, A. K., Giupponi, C., & Wada, Y. (2016). Measuring global water security towards sustainable development goals. *Environmental Research Letters*, *11*(12), 124015. https://doi.org/10.1088/1748-9326/11/12/124015

Galatowitsch, S. M. (2012). *Ecological Restoration*. Sunderland, USA: Sinauer Associates.

García del Valle, Y., Naranjo, E. J., Caballero, J., Martorell, C., Ruan-Soto, F., & Enríquez, P. L. (2015). Cultural significance of wild mammals in mayan and mestizo communities of the Lacandon Rainforest, Chiapas, Mexico. *Journal of Ethnobiology and Ethnomedicine*, *11*(36), (pp. 1–13). https://doi.org/10.1186/s13002-015-0021-7

Garibaldi, L., & Muchhala, N. (2011). Services from Plant–Pollinator Interactions in the Neotropics. In B. Rapidel, D. F., L. C. J. F., & B. J. (Eds.), *Ecosystem services from agriculture and agroforestry: measurement and payment* (pp. 119–140). Earthscan. https://doi.org/10.4324/9781849775656

Garner, K., & Parfitt, B. (2006). *First Nations, Salmon Fisheries and the Rising Importance of Conservation*. Vancouver, BC.

GCDT. (2007). *Global Strategy for the Ex Situ Conservation and Use of Barley Germplasm*. Rome.

Geist, H., McConnell, W., Lambin, E. F., Moran, E., Alves, D., & Rudel, T. (2006). Causes and Trajectories of Land-Use/Cover Change. In E. F. Lambin & H. Geist (Eds.), *Land-Use and Land-Cover Change: Local Processes and Global Impacts* (pp. 41–70). Berlin, Heidelberg: Springer Berlin Heidelberg. https://doi.org/10.1007/3-540-32202-7_3

Gerber, P. J., Steinfeld, H., Henderson, B., Mottet, A., Opio, C., Dijkman, J., Falcucci, A. & Tempio, G. (2013). *Tackling climate change through livestock – A global assessment of emissions and mitigation opportunities*. Rome, Italy: Food and Agriculture Organization of the United Nations (FAO). https://doi.org/10.1016/j.anifeedsci.2011.04.074

Giannini, T. C., Cordeiro, G. D., Freitas, B. M., Saraiva, a. M., & Imperatriz-Fonseca, V. L. (2015). The Dependence of Crops for Pollinators and the Economic Value of Pollination in Brazil. *Journal of Economic Entomology*, *108*(3), (pp. 1–9). https://doi.org/10.1093/jee/tov093

GIZ – Deutsche Gesellschaft für Internationale Zusammenarbeit. (2014). *Multiple-Household Fuel Use – a balanced choice between firewood, charcoal and LPG*. Bonn, Germany. Retrieved from http://www.eco-consult.com/fileadmin/user_upload/pdf/Multiple-Household_Fuel_Use.pdf

Gleick, P. (2014). *The World's Water Volume 8: The Biennial Report on Freshwater Resources*. Washington DC: Island Press.

Global Food Print Network. (2016). *National Footprint Accounts (2016 Edition).*

Gobierno Departamental Autónomo de Santa Cruz. (2009). *Propuesta para el manejo de fauna silvestre y lineamientos para promover el biocomercio en el Departamento de Santa Cruz*. Santa Cruz, Bolivia.

Godfray, H. C. J., Beddington, J. R., Crute, I. R., Haddad, L., Lawrence, D., Muir, J. F., Pretty, J., Robinson, S., Thomas, S. M., & Toulmin, C. (2010). Food security: the challenge of feeding 9 billion people. *Science (New York, N.Y.)*, *327*(5967), 812–818. https://doi.org/10.1126/science.1185383

Godoy, R., Reyes-García, V., Byron, E., Leonard, W. R., & Vadez, V. (2005). The Effect of Market Economies on the Well-Being of Indigenous Peoples and on their Use of Renewable Natural Resources. *Annual Review of Anthropology*, *34*(1), (pp. 121–138). https://doi.org/10.1146/annurev.anthro.34.081804.120412

Goijman AP, Conroy MJ, Bernardos JN, Zaccagnini ME (2015) Multi-Season Regional Analysis of Multi-Species Occupancy: Implications for Bird Conservation in Agricultural Lands in East-Central Argentina. PLoS ONE 10(6): e0130874. https://doi.org/10.1371/journal.pone.0130874

Goijman, A. & Zaccagnini, M.E. (2008). The effects of habitat heterogeneity on avian density and richness in soybean field in Entre Ríos, Argentina. El Hornero 23(2):67-76

Gómez-Baggethun, E., de Groot, R., Lomas, P. L., & Montes, C. (2010). The history of ecosystem services in economic theory and practice: From early notions to markets and payment schemes. *Ecological Economics*, *69*(6), (pp. 1209–1218). https://doi.org/https://doi.org/10.1016/j.ecolecon.2009.11.007

González-de la V., M. (2004). *Cacao: 1500 años en Tabasco*. *Arqueología Mexicana-Gobierno Del Estado*, 74–77.

Gorte, R. (2013). *The Rising Cost of Wildfire Protection*. *Headwaters Economics*, (June), 19. https://doi.org/http://headwaterseconomics.org/wildfire/fire-costs-background

Graeub, B. E., Chappell, M. J., Wittman, H., Ledermann, S., Kerr, R. B., & Gemmill-Herren, B. (2016). The State of Family Farms in the World. *World Development*, *87*, (pp. 1–15). https://doi.org/10.1016/j.worlddev.2015.05.012

Graham-Rowe, D. (2011). *Biodiversity: Endangered and in demand*. *Nature*, *480*(7378), S101–S103. Retrieved from http://dx.doi.org/10.1038/480S101a

Great Lakes and Mississippi River Interbasin Study Team (GLMRIST). (2012). *Commercial Fisheries Baseline Economic Assessment - U.S. Waters of the Great Lakes, Upper Mississippi River, and Ohio River Basins*. Detroit, MI: U.S. Army Corps of Engineers.

Grey, D., & Sadoff, C. W. (2007). Sink or Swim? Water security for growth and development. *Water Policy*, *9*(6), (pp. 545). https://doi.org/10.2166/wp.2007.021

Grigal, D. F. (2003). *Mercury sequestration in forests and peatlands: a review. Journal of Environmental Quality*, *32*(2), (pp. 393–405).

Grigg, N. S. (2005). *Water Resources Management. In Water Encyclopedia* (p. 2:586–587.). John Wiley & Sons, Inc. https://doi.org/10.1002/047147844X.wr241

Grimm, N. B., Foster, D., Groffman, P., Grove, J. M., Hopkinson, C. S., Nadelhoffer, K. J., Pataki, D. E., & Peters, D. P. (2008). The changing landscape: ecosystem responses to urbanization and pollution across climatic and societal gradients. *Frontiers in Ecology and the Environment*, *6*(5), 264–272. https://doi.org/10.1890/070147

Grimm, N. B., Groffman, P., Staudinger, M., & Tallis, H. (2016). Climate change impacts on ecosystems and ecosystem services in the United States: process and prospects for sustained assessment. *Climatic Change*, *135*(1), (pp. 97–109). https://doi.org/10.1007/s10584-015-1547-3

Grinde, B., & Patil, G. G. (2009). Biophilia: Does visual contact with nature impact on health and well-being? *International Journal of Environmental Research and Public Health*, *6*(9), (pp. 2332–2343). https://doi.org/10.3390/ijerph6092332

Guo, L. B., & Gifford, R. M. (2002). Soil carbon stocks and land use change: a meta analysis. *Global Change Biology*, *8*(4), (pp. 345–360). https://doi.org/10.1046/j.1354-1013.2002.00486.x

Haddad, N. (2000). *Corridor Length and Patch Colonization by a Butterfly, Junonia coenia. Conservation Biology, 14(3),* (pp. 738–745). https://doi.org/10.1046/j.1523-1739.2000.99041.x

Haddad, N. M., Brudvig, L. A., Clobert, J., Davies, K. F., Gonzalez, A., Holt, R. D., Lovejoy, T. E., Sexton, J. O., Austin, M. P., Collins, C. D., Cook, W. M., Damschen, E. I., Ewers, R. M., Foster, B. L., Jenkins, C. N., King, A. J., Laurance, W. F., Levey, D. J., Margules, C. R., Melbourne, B. A., Nicholls, A. O., Orrock, J. L., Song, D.-X., & Townshend, J. R. (2015). Habitat fragmentation and its lasting impact on Earth's ecosystems. *Science Advances*, *1*(2), e1500052–e1500052. https://doi.org/10.1126/sciadv.1500052

Halls, L. K. (1984). *White Tailed Deer: Ecology and Management*. (R. E. McCabe & R. J. Laurence, Eds.) (Wildlife M). Washington, D.C.: Stackpole Books.

Hamilton, L. S. (2008). *Forest and Water: A thematic study prepared in the framework of the Global Forest Resources Assessment 2005* (FAO Forestry Paper No. 155). Rome.

Hanasaki, N., Fujimori, S., Yamamoto, T., Yoshikawa, S., Masaki, Y., Hijioka, Y., Kainuma, M., Kanamori, Y., Masui, T., Takahashi, K., & Kanae, S. (2013). A global water scarcity assessment under Shared Socio-economic Pathways – Part 2: Water availability and scarcity. *Hydrology and Earth System Sciences*, *17*(7), 2393–2413. https://doi.org/10.5194/hess-17-2393-2013

Hanley, N., & Spash, C. L. (1993). *Cost-benefit analysis and the environment*. Northampton, MA: Edward Algar.

Harr, R. D. (1986). Effects of Clearcutting on Rain-on-Snow Runoff in Western Oregon: A New Look at Old Studies. *Water Resources Research*, *22*(7), (pp. 1095–1100). https://doi.org/10.1029/WR022i007p01095

Harris, J. M., & Roach, B. (2014). *Environmental and Natural Resource Economics: A Contemporary Approach* (3rd Editio). New York, NY: Routledge.

Hartig, T., Mang, M., & Evans, G. W. (1991). Restorative Effects of Natural Environment Experiences. *Environment and Behavior*, *23*(1), (pp. 3–26). https://doi.org/10.1177/0013916591231001

Hasson, F., Keeney, S., & McKenna, H. (2000). Research guidelines for the Delphi survey technique. *Journal of Advanced Nursing*, *32*(4), (pp. 1008–1015). https://doi.org/10.1046/j.1365-2648.2000.t01-1-01567.x

Hellin, J., Erenstein, O., Beuchelt, T., Camacho, C., & Flores, D. (2013). Maize stover use and sustainable crop production in mixed crop-livestock systems in Mexico. *Field Crops Research*, *153*, (pp. 12–21). https://doi.org/10.1016/j.fcr.2013.05.014

Herendeen, R. A., & Wildermuth, T. (2002). Resource-based sustainability indicators: Chase County, Kansas, as example. *Ecological Economics*, *42*(1–2), (pp. 243–257). https://doi.org/https://doi.org/10.1016/S0921-8009(02)00056-3

Hervé, D., & Ayangma, S. (2000). Dynamique de l'occupation du sol dans une communauté agro-pastorale de'l Altiplano bolivien. Les montagnes aménagement. *Revue de Géographie Alpine*, *88*(2), (pp. 69–84).

Herzog, S., Martínez, R., Jorgensen, P., & Tiessen, H. (2011). *Climate Change and Biodiversity in the Tropical Andes*. *Work*. Inter-American Institute for Global Change Research (IAI) and Scientific Committee on Problems of the Environment (SCOPE).

Hilbeck, A., Binimelis, R., Defarge, N., Steinbrecher, R., Székács, A., Wickson, F., Antoniou, M., Bereano, P. L., Clark, E. A., Hansen, M., Novotny, E., Heinemann, J., Meyer, H., Shiva, V., & Wynne, B. (2015). No scientific consensus on GMO safety. *Environmental Sciences Europe*, *27*(1), 4. https://doi.org/10.1186/s12302-014-0034-1

Hilty, J. A., & Merenlender, A. M. (2004). Use of Riparian Corridors and Vineyards by Mammalian Predators in Northern California. *Conservation Biology*, *18*(1), (pp. 126–135). https://doi.org/10.1111/j.1523-1739.2004.00225.x

Hoekstra, A. Y., Mekonnen, M. M., Chapagain, A. K., Mathews, R. E., & Richter, B. D. (2012). Global monthly water scarcity: Blue water footprints versus blue water availability. *PLoS ONE*, *7*(2). https://doi.org/10.1371/journal.pone.0032688

Hoffmann, I., From, T., & Boerma, D. (2014). *Ecosystem servicies provided by livestock species and breeds, with special consideration to the contributions of small-scale livestock keepers and pastoralists*. Rome, Italy. https://doi.org/10.1146/annurev.energy.29.062403.102203

Hollaway, M. J., Arnold, S. R., Challinor, A. J., & Emberson, L. D. (2012). Intercontinental trans-boundary contributions to ozone-induced crop yield losses in the Northern Hemisphere. *Biogeosciences*, *9*(1), (pp. 271–292). https://doi.org/10.5194/bg-9-271-2012

Honda, E. A., Durigan, G., Bowman, D., Walsh, A., Milne, D., Roques, K., … Midgley, G. (2016). Woody encroachment and its consequences on hydrological processes in the savannah. *Philosophical Transactions of the Royal Society of London. Series B, Biological Sciences*, *371*(1703), 115–133. https://doi.org/10.1098/rstb.2015.0313

Horowitz, L. W. (2006). *Past, present and future concentrations of tropospheric ozone and aerosols: Methodology, ozone evaluation, and sensitivity to aerosol wet removal. Journal of Geophysical Research Atmospheres, 111(22)*. https://doi.org/10.1029/2005JD006937

Horwitz, P., Finlayson, C. M., & Weinstein, P. (2012). *Healthy wetlands, healthy people: a review of wetlands and human health interactions. Ramsar Technical Reports*. Gland, Switzerland. Retrieved from http://www.ramsar.org/sites/default/files/documents/pdf/lib/rtr6-health.pdf

Houston, D. B. (1982). *The Northern Yellowstone Elk: Ecology and Management*. Macmillan.

Howe, C., Suich, H., Vira, B., & Mace, G. M. (2014). Creating win-wins from trade-offs? Ecosystem services for human well-being: A meta-analysis of ecosystem service trade-offs and synergies in the real world. *Global Environmental Change*, *28*, 263–275. https://doi.org/10.1016/j.gloenvcha.2014.07.005

Huang, Z. Y. X., Van Langevelde, F., Estrada-Peña, A., Suzán, G., & De Boer, W. F. (2016). The diversity–disease relationship: evidence for and criticisms of the dilution effect. *Parasitology*, *143*(9), (pp. 1075–1086). https://doi.org/10.1017/S0031182016000536

Huddleston, B., Ataman, E., Salvo, S. P. De, Zanetti, M., Bloise, M., Bel, J., Franceschini, G., & D'Ostiani, L. F. (2003). *Towards a Gis-Based Analysis of Mountain*. *Environment and Natural Resources Working Paper No. 10*.

Hummer, K. E., & Hancock, J. F. (2015). HortScience : a publication of the American Society for Horticultural Science. *HortScience*, 50(6), 780–783.

IDB. (2016). *Inter-american Development Bank.* Retrieved from http://www.iadb.org/en/research-and-data/research-publications,19532.html

IMF. (2015). *Data file of world economic and financial surveys*. Retrieved August 15, 2017, from https://www.imf.org/external/pubs/ft/weo/2015/02/weodata/index.aspx

Immirzi, C. P., Maltby, E., & Clymo, R. S. (1992). *The global status of peatlands and their role in carbon cycling: a report for Friends of the Earth*. Friends of the Earth, Wetland Ecosystems Research Group.

International Association of Fishing and Wildlife Agencies (IAFWA). (2002). *The Economic Importance of Hunting in America*. (I. A. of F. and W. Agencies, Ed.).

International Energy Agency (IEA) and the World Bank. (2006). *World Energy Outlook 2006*. *Outlook*. https://doi.org/10.1787/weo-2006-en

International Energy Agency (IEA) and the World Bank. (2017). *Sustainable Energy for All 2017—Progress toward Sustainable Energy*. Washington, D.C.

IPBES. (2016). *Summary for policymakers of the assessment report of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services on pollinators, pollination and food production.* (N. G. S.G. Potts, V. L. Imperatriz-Fonseca, H. T. Ngo, J. C. Biesmeijer, T. D. Breeze, L. V. Dicks, L. A. Garibaldi, R. Hill, J. Settele, A. J. Vanbergen, M. A. Aizen, S. A. Cunningham, C. Eardley, B. M. Freitas & and B. F. V. (eds.). P. G. Kevan, A. Kovács-Hostyánszki, P. K. Kwapong, J. Li, X. Li, D. J. Martins, G. Nates-Parra, J. S. Pettis, R. Rader, and B. F. Viana (edR. Rader, Eds.). Bonn, Germany: Secretariat of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services.

IPCC. (2001). *Climate change 2001: impacts, adaptation and vulnerability, Contribution of Working Group II to the Third Assessment Report of the Intergovernmental Panel on Climate Change*. (D. J. D. and K. S. W. J. McCarthy, O. F. Canziani, N. A. Leary, Ed.). Cambridge, UK and New York, USA: Cambridge University Press.

IPCC. (2007). *Climate change 2007: Impacts, adaptation and vulnerability*. (M. L. Parry, O. F. Canziani, J. P. Palutikof, P. J. van der Linden, & C. E. Hanson, Eds.). Cambridge, UK: Cambridge University Press.

Isaac, V. J., Almeida, M. C., Cruz, R. E. A., & Nunes, L. G. (2015). Artisanal fisheries of the Xingu River basin in Brazilian Amazon. *Brazilian Journal of Biology*, *75*(3 Suppl 1), (pp. 125–37). https://doi.org/10.1590/1519-6984.00314BM

Islam, D., & Berkes, F. (2016). Indigenous peoples' fisheries and food security: a case from northern Canada. *Food Security*, *8*(4), (pp. 815–826). https://doi.org/10.1007/s12571-016-0594-6

Iwamura, T., Lambin, E. F., Silvius, K. M., Luzar, J. B., & Fragoso, J. M. V. (2014). Agent-based modeling of hunting and subsistence agriculture on indigenous lands: Understanding interactions between social and ecological systems. *Environmental Modelling and Software*, *58*, (pp. 109–127). https://doi.org/10.1016/j.envsoft.2014.03.008

Jackson, J. B. C., Donovan, M. K., Cramer, K. L., Lam, V., & Lam, W. (editors). (2014). *Status and Trends of Caribbean Coral Reefs : 1970-2012*. *Global Coral Reef Monitoring Network, IUCN, Gland, Switzerland*.

Jackson, J. B., Kirby, M. X., Berger, W. H., Bjorndal, K. A., Botsford, L. W., Bourque, B. J., Bradbury, R. H., Cooke, R., Erlandson, J., Estes, J. A., Hughes, T. P., Kidwell, S., Lange, C. B., Lenihan, H. S., Pandolfi, J. M., Peterson, C. H., Steneck, R. S., Tegner, M. J., & Warner, R. R. (2001). Historical overfishing and the recent collapse of coastal ecosystems. *Science (New York, N.Y.)*, *293*(5530), 629–637. https://doi.org/10.1126/science.1059199

Jackson, R. B. (2005). *Trading Water for Carbon with Biological Carbon Sequestration. Science, 310(5756)*, (pp. 1944–1947). https://doi.org/10.1126/science.1119282

Janick, J. (2013). Development of new world crops by indigenous Americans. *HortScience*, *48*(4), 406–412.

Jarvis, A., Lane, A., & Hijmans, R. J. (2008). The effect of climate change on crop wild relatives. *Agriculture, Ecosystems and Environment*, *126*(1–2), (pp. 13–23). https://doi.org/10.1016/j.agee.2008.01.013

Jenerette, G. D., Harlan, S. L., Stefanov, W. L., & Martin, C. A. (2011). Ecosystem services and urban heat riskscape moderation: water, green spaces, and social inequality in Phoenix, USA. *Ecological Applications*, *21*(7), (pp. 2637–2651). https://doi.org/10.1890/10-1493.1

Jenkins, W. A., Murray, B. C., Kramer, R. A., & Faulkner, S. P. (2010). Valuing ecosystem services from wetlands restoration in the Mississippi Alluvial Valley. *Ecological Economics*, *69*(5), (pp. 1051–1061). https://doi.org/https://doi.org/10.1016/j.ecolecon.2009.11.022

Johnson, P. T. J., & Thieltges, D. W. (2010). Diversity, decoys and the dilution effect: how ecological communities affect disease risk. *Journal of Experimental Biology*, *213*(6), (pp. 961–970). https://doi.org/10.1242/jeb.037721

Joly, C., & Bolzani, V. (2016). The Challenge of Including Chemodiversity, and the Potential Economic Use of New Natural Compounds and Processes, in the BIOTA/FAPESP Program. *Journal of the Brazilian Chemical Society*. https://doi.org/10.21577/0103-5053.20160320

Jones, K. E., Patel, N. G., Levy, M. A., Storeygard, A., Balk, D., Gittleman, J. L., & Daszak, P. (2008). Global trends in emerging infectious diseases. *Nature*, *451*, (pp. 990). Retrieved from http://dx.doi.org/10.1038/nature06536

Jones, R. (2013). Stingless Bees: A Historical Perspective. In *Pot-Honey: A Legacy of Stingless Bees* (pp. 1–654). New York: Springer Science+Business Media Dordrecht. https://doi.org/10.1007/978-1-4614-4960-7

Jones-Walters, L., & Çil, A. (2011). Biodiversity and stakeholder participation. *Journal for Nature Conservation*, *19*(6), (pp. 327–329). https://doi.org/10.1016/j.jnc.2011.09.001

Joppa, L. N., Loarie, S. R., & Pimm, S. L. (2008). On the protection of "protected areas". *Proceedings of the National Academy of Sciences*, *105*(18), 6673–8. https://doi.org/10.1073/pnas.0802471105

Jorgenson, J. P. (1993). *Gardens, wildlife densities, and subsistence hunting by maya indians in Quintana Roo, Mexico.* Gainesville: University of Florida.

Kaimowitz, D., Fauné, A., & Mendoza, R. (2003). Your biosphere is my backyard: The story of BOSAWAS in Nicaragua. *Policy Matters*, (pp. 6–15).

Kalin-Arroyo, M. T., Dirzo, R., Joly, C. A., Castilla, J. C., & Rodrigues, F. C. (2009). *Biodiversity knowledge, scope of research and priority areas: an assessment of Latin America and the Caribbean*. Rio de Janeiro, Brazil: ICSU-LAC.

Kaplan, R. (2001). *The Nature of the View from Home: Psychological Benefits. Environment and Behavior, 33(4*), (pp. 507–542). https://doi.org/10.1177/00139160121973115

Kapos, V., Balmford, A., Aveling, R., Bubb, P., Carey, P., Entwistle, A., Hopkins, J., Mulliken, T., Safford, R., Stattersfield, A., Walpole, M., & Manica, A. (2008). Calibrating conservation: new tools for measuring success. *Conservation Letters*, *1*(4), 155–164. https://doi.org/10.1111/j.1755-263X.2008.00025.x

Karlen, D., & Rice, C. (2015). Soil Degradation: Will Humankind Ever Learn? *Sustainability*, *7*(9), (pp. 12490–12501). https://doi.org/10.3390/su70912490

Kay, K. M., Reeves, P. A., Olmstead, R. G., & Schemske, D. W. (2005). Rapid speciation and the evolution of hummingbird pollination in neotropical Costus subgenus Costus (Costaceae): evidence from nrDNA ITS and ETS sequences. *American Journal of Botany*, *92*(11), 1899–1910. https://doi.org/10.3732/ajb.92.11.1899

Kearney, J. (2010). *Food consumption trends and drivers*. *Philosophical Transactions of the Royal Society B: Biological Sciences*, *365*(1554), (pp. 2793–2807). https://doi.org/10.1098/rstb.2010.0149

Keen, M., Brown, V. A., & Dyball, R. (2005). *Social learning in environmental management : towards a sustainable future*. Sterling, Va: Earthscan.

Keniger, L. E., Gaston, K. J., Irvine, K. N., & Fuller, R. A. (2013). What are the benefits of interacting with nature? *International Journal of Environmental Research and Public Health*, *10*(3), (pp. 913–935). https://doi.org/10.3390/ijerph10030913

Kenny, J. F., Barber, N. L., Hutson, S. S., Linsey, K. S., Lovelace, J. K., & Maupin, M. A. (2009). *Estimated use of water in the United States in 2005: U.S. Geological Survey Circular 1344*. Reston, VA: U.S. Geological Survey.

Kessler, M. (2001). *Patterns of diversity and range size of selected plant groups along an elevational transect in the Bolivian Andes. Biodiversity & Conservation*, *10*(11), (pp. 1897–1921). https://doi.org/10.1023/A:1013130902993

Khan, I., Yang, Z., Maldonado, E., Li, C., Zhang, G., Gilbert, M. T. P., Jarvis, E. D., O'Brien, S. J., Johnson, W. E., & Antunes, A. (2015). Olfactory Receptor Subgenomes Linked with Broad Ecological Adaptations in Sauropsida. *Molecular Biology and Evolution*, *32*(11), 2832–2843. https://doi.org/10.1093/molbev/msv155

Khan, S., Ahmad, I., Shah, M. T., Rehman, S., & Khaliq, A. (2009). Use of constructed wetland for the removal of heavy metals from industrial wastewater. *Journal of Environmental Management*, *90*(11), (pp. 3451–3457). https://doi.org/10.1016/j.jenvman.2009.05.026

Kim, C., Scott, D., Thigpen, J., & Kim, S.-S. (1998). Economic Impact of a Birding Festival. *Festival Management & Event Tourism*, *5*(July), (pp. 51–58). https://doi.org/10.3727/106527098792186702

Kim, J. H., Lee, C., & Sohn, W. (2016). Urban natural environments, obesity, and health-related quality of life among hispanic children living in Inner-City neighborhoods. *International Journal of Environmental Research and Public Health*, *13*(1). https://doi.org/10.3390/ijerph13010121

Kirmayer, L. J., Brass, G. M., & Tait, C. L. (2000). The Mental Health of Aboriginal Peoples: Transformations of Identity and Community. *The Canadian Journal of Psychiatry*, *45*(7), (pp. 607–616). https://doi.org/10.1177/070674370004500702

Klain, S. C., & Chan, K. M. A. (2012). Navigating coastal values: Participatory mapping of ecosystem services for spatial planning. *Ecological Economics*, *82*, (pp. 104–113). https://doi.org/10.1016/j.ecolecon.2012.07.008

Kleypas, J. A., Feely, R. A., Fabry, V. J., Langdon, C., Sabine, C. L., & Robbins, L. L. (2006). *Impacts of Ocean Acidification on Coral Reefs and Other Marine Calcifiers: A Guide for Future Research*. *A report of a workshop held 18–20 April 2005, St. Petersburg, FL, sponsored by NSF, NOAA, and the U.S. Geological Survey*. https://doi.org/www.ucar.edu/communications/Final_acidification.pdf

Koizumi, T. (2014). *Biofuels and Food Security*. *Cham: Springer International Publishing*. https://doi.org/10.1007/978-3-319-05645-6

Kollmann, J., Meyer, S. T., Bateman, R., Conradi, T., Gossner, M. M., de Souza Mendonça, M., Fernandes, G. W., Hermann, J.-M., Koch, C., Müller, S. C., Oki, Y., Overbeck, G. E., Paterno, G. B., Rosenfield, M. F., Toma, T. S. P., & Weisser, W. W. (2016). Integrating ecosystem functions into restoration ecology-recent advances and future directions. *Restoration Ecology*, *24*(6), 722–730. https://doi.org/10.1111/rec.12422

Konikow, L. F. (2013). *Groundwater Depletion in the United States (1900–2008). Scientific Investigations Report 2013–5079*. Reston, Virginia.

Krapovickas, A. (2010). *The domestication and origin of agriculture*. *Bonplandia*, *19*(2), (pp. 193–199).

Krömer, T., Kessler, M., & Herzog, S. K. (2006). Distribution and Flowering Ecology of Bromeliads along Two Climatically Contrasting Elevational Transects in the Bolivian Andes1. *Biotropica*, *38*(2), (pp. 183–195). https://doi.org/10.1111/j.1744-7429.2006.00124.x

Kubiszewski, I., Costanza, R., Anderson, S., & Sutton, P. (2017). The future value of ecosystem services: Global scenarios and national implications. *Ecosystem Services*, *26*, (pp. 289–301). https://doi.org/10.1016/j.ecoser.2017.05.004

Kuhnlein, H. V, & Chan, H. M. (2000). Environment and contaminants in traditional food systems of northern indigenous peoples. *Annual Review of Nutrition*, *20*(1), (pp. 595–626). https://doi.org/10.1146/annurev.nutr.20.1.595

Kuhnlein, H. V., & Humphries, M. M. (2017). Traditional Animal Foods of Indigenous Peoples of Northern North America. Retrieved March 3, 2017, from http://traditionalanimalfoods.org/

Kuipers, S. E. (1997). Trade in Medicinal Plants. In G. Bodeker, K. K. S. Bhat, J. Burley, & P. Vantomme (Eds.), *Medicinal plants for forest conservation and health care* (pp. 45–59). Rome, FAO.: FAO.

Kujawska, M., Hilgert, N. I., Keller, H. A., & Gil, G. (2017). Medicinal Plant Diversity and Inter-Cultural Interactions between Indigenous Guarani, Criollos and Polish Migrants in the Subtropics of Argentina. *Plos One*, *12*(1), e0169373. https://doi.org/10.1371/journal.pone.0169373

Kushner, B., Edwards, P., Burke, L., & Cooper, E. (2011). *Coastal Capital: Jamaica. Coral Reefs, Beach Erosion and Impacts to Tourism in Jamaica.* Washington, D.C. Retrieved from http://www.wri.org/coastal-capital.

Labate, B. C. (2004). *A reinvenção do uso da ayahuasca nos centros urbanos*. Editora Mercado de Letras.

Ladio, A. H. (2011). Underexploited wild plant foods of North-Western Patagonia. *Multidisciplinary Approaches on Food Science and Nutrition for the XXI Century*, *661*(2), (pp. 1–16). Retrieved from http://scholar.google.com/scholar?hl=en&btnG=Search&q=intitle:Underexploited+wild+plant+foods+of+north-western+patagonia#0

Laird, S. (1999). *The botanical medicine industry. In K. Ten Kate & S. A. Laird (Eds.), The Commercial Use of Biodiversity: Access to Genetic Resources and Benefit Sharing* (pp. 78–116). London: Earthscan.

Laird, S. A., & Pierce, A. R. (2002). *Promoting sustainable and ethical botanicals. Strategies to improve commercial raw material sourcing. Results from the sustainable botanicals pilot project. Industry surveys, case studies and standards collection.Industry surveys, case studies and standar*. New York, NY.

Laird, S., & ten Kate, K. (1999). Natural products and the pharmaceutical industry. In K. ten Kate & S. Laird (Eds.), *The Commercial Use of Biodiversity: Access to Genetic Resources and Benefit Sharing* (pp. 34–77). London: Earthscan.

Lal, R. (2004). Soil carbon sequestration to mitigate climate change. *Geoderma*, *123*(1–2), 1–22.

https://doi.org/10.1016/j.geoderma.2004.01.032

Lal, R. (2008). *Carbon sequestration*. *Philos. Trans. R. Soc., B*, *363*(August 2007), (pp. 815–830). https://doi.org/10.1098/rstb.2007.2185

Lal, R., Kimble, J., Levine, E., & Stewart. (1995). *Soils and Global Change*. CRC Press.

Lamb, D., Erskine, P. D., & Parrotta, J. A. (2005). Restoration of Degraded Tropical Forest Landscapes. *Science*, *310*(5754), (pp. 1628–1632). https://doi.org/10.1126/science.1111773

Lambertucci, S. A., Donázar, J. A., Huertas, A. D., Jiménez, B., Sáez, M., Sanchez-Zapata, J. A., & Hiraldo, F. (2011). Widening the problem of lead poisoning to a South-American top scavenger: Lead concentrations in feathers of wild Andean condors. *Biological Conservation*, *144*(5), (pp. 1464–1471). https://doi.org/10.1016/j.biocon.2011.01.015

Lambin, E.F., Turner, B.L., Geist, H.J., Agbola, S.B., Angelsen, A., Bruce, J.W., Coomes, O. T., Dirzgo, R., Fischer, G., Folke, C., George, P.S., Homewood, K., Imbernon, J., Leemans, R., Li, X., Moran, E.F., Mortimore, M., Ramakrishnan, P.S., Richards, J.F., Skanes, H., Steffen, W., & Stone, G.D., Svedin, U., Veldkamp, T.A., Vogel, C., and Xu, J. (2001). The Causes of Land-Use and Land-Cover Change: Moving Beyond the Myths. *Global Environmental Change*, *11*, (pp. 261–269).

Landeta, J. (2006). *Current validity of the Delphi method in social sciences. Technological Forecasting and Social Change*, *73*(5), (pp. 467–482). https://doi.org/10.1016/j.techfore.2005.09.002

Landis, D. A. (2017). *Designing agricultural landscapes for biodiversity-based ecosystem services. Basic and Applied Ecology, 18*, (pp. 1–12). https://doi.org/10.1016/j.baae.2016.07.005

Lange, D. (1998). *Europe's Medicinal and Aromatic Plants: Their Use, Trade and Conservation*. Cambridge, UK: TRAFFIC International.

Laterra, P., Jobbágy, E., & Paruelo, J. M. (2011). *Valoración de Servicios Ecosistémicos: Conceptos, Herramientas y Aplicaciones Para el Ordenamiento Territorial*. (P. Laterra, E. Jobbágy, & J. M. Paruelo, Eds.), *Valoración de Servicios Ecosistémicos: Conceptos, Herramientas y Aplicaciones Para el Ordenamiento Territorial*.

Lavery, P. S., Mateo, M. Á., Serrano, O., & Rozaimi, M. (2013). Variability in the Carbon Storage of Seagrass Habitats and Its Implications for Global Estimates of Blue Carbon Ecosystem Service. *PLoS ONE*, *8*(9). https://doi.org/10.1371/journal.pone.0073748

Lawler, J. J., Lewis, D. J., Nelson, E., Plantinga, A. J., Polasky, S., Withey, J. C., Helmers, D. P., Martinuzzi, S., Pennington, D., & Radeloff, V. C. (2014). Projected land-use change impacts on ecosystem services in the United States. *Proceedings of the National Academy of Sciences*, *111*(20), 7492–7497.

Layke, C. (2009). *Measuring Nature ' s Benefits : A Preliminary Roadmap for Improving Ecosystem Service Indicators. Analysis, World Reso*, (pp. 1–36). Retrieved from http://www.wri.org/project/

Le Quéré, C., Takahashi, T., Buitenhuis, E. T., Rödenbeck, C., & Sutherland, S. C. (2010). Impact of climate change and variability on the global oceanic sink of CO2. *Global Biogeochemical Cycles*, *24*(4), (pp. 1–10). https://doi.org/10.1029/2009GB003599

Leader-Williams, N., Adams, W. M., & Smith, R. J. (2010). *Tradeoffs in Conservation: Deciding What to Save*. Hoboken, NJ, USA: John Wiley & Sons.

Lehner, B., & Döll, P. (2004). Development and validation of a global database of lakes, reservoirs and wetlands. *Journal of Hydrology*, *296*(1–4), (pp. 1–22). https://doi.org/10.1016/j.jhydrol.2004.03.028

Leverington, F., Costa, K. L., Pavese, H., Lisle, A., & Hockings, M. (2010). A Global Analysis of Protected Area Management Effectiveness. *Environmental Management*, *46*(5), (pp. 685–698). https://doi.org/10.1007/s00267-010-9564-5

Levetin, E., & McMahon, K. (2008). Materials: Cloth, Wood, and Paper. In *Plants and Society* (Fifth Edit, pp. 298–323). The McGraw-Hill Companies.

Levis, C., Costa, F. R. C., Bongers, F., Peña-Claros, M., Clement, C. R., Junqueira, A. B., Neves, E. G., Tamanaha, E. K., Figueiredo, F. O. G., Salomão, R. P., Castilho, C. V, Magnusson, W. E., Phillips, O. L., Guevara, J. E., Sabatier, D., Molino, J.-F., López, D. C., Mendoza, A. M., Pitman, N. C. A., Duque, A., Vargas, P. N., Zartman, C. E., Vasquez, R., Andrade, A., Camargo, J. L., Feldpausch, T. R., Laurance, S. G. W., Laurance, W. F., Killeen, T. J., Nascimento, H. E. M., Montero, J. C., Mostacedo, B., Amaral, I. L., Guimarães Vieira, I. C., Brienen, R., Castellanos, H., Terborgh, J., Carim, M. de J.

V., Guimarães, J. R. da S., Coelho, L. de S., Matos, F. D. de A., Wittmann, F., Mogollón, H. F., Damasco, G., Dávila, N., García-Villacorta, R., Coronado, E. N. H., Emilio, T., Filho, D. de A. L., Schietti, J., Souza, P., Targhetta, N., Comiskey, J. A., Marimon, B. S., Marimon, B.-H., Neill, D., Alonso, A., Arroyo, L., Carvalho, F. A., de Souza, F. C., Dallmeier, F., Pansonato, M. P., Duivenvoorden, J. F., Fine, P. V. A., Stevenson, P. R., Araujo-Murakami, A., Aymard C, G. A., Baraloto, C., do Amaral, D. D., Engel, J., Henkel, T. W., Maas, P., Petronelli, P., Revilla, J. D. C., Stropp, J., Daly, D., Gribel, R., Paredes, M. R., Silveira, M., Thomas-Caesar, R., Baker, T. R., da Silva, N. F., Ferreira, L. V, Peres, C. A., Silman, M. R., Cerón, C., Valverde, F. C., Di Fiore, A., Jimenez, E. M., Mora, M. C. P., Toledo, M., Barbosa, E. M., Bonates, L. C. de M., Arboleda, N. C., Farias, E. de S., Fuentes, A., Guillaumet, J.-L., Jørgensen, P. M., Malhi, Y., de Andrade Miranda, I. P., Phillips, J. F., Prieto, A., Rudas, A., Ruschel, A. R., Silva, N., von Hildebrand, P., Vos, V. A., Zent, E. L., Zent, S., Cintra, B. B. L., Nascimento, M. T., Oliveira, A. A., Ramirez-Angulo, H., Ramos, J. F., Rivas, G., Schöngart, J., Sierra, R., Tirado, M., van der Heijden, G., Torre, E. V, Wang, O., Young, K. R., Baider, C., Cano, A., Farfan-Rios, W., Ferreira, C., Hoffman, B., Mendoza, C., Mesones, I., Torres-Lezama, A., Medina, M. N. U., van Andel, T. R., Villarroel, D., Zagt, R., Alexiades, M. N., Balslev, H., Garcia-Cabrera, K., Gonzales, T., Hernandez, L., Huamantupa-Chuquimaco, I., Manzatto, A. G., Milliken, W., Cuenca, W. P., Pansini, S., Pauletto, D., Arevalo, F. R., Reis, N. F. C., Sampaio, A. F., Giraldo, L. E. U., Sandoval, E. H. V., Gamarra, L. V., Vela, C. I. A., & Ter Steege, H. (2017). Persistent effects of pre-Columbian plant domestication on Amazonian forest composition. *Science (New York, N.Y.)*, *355*(6328), 925–931. https://doi.org/10.1126/science.aal0157

Lichtenstein, G., & Vilá, B. (2003). Vicuna Use by Andean Communities: An Overview. *Mountain Research and Development*, *23*(2), (pp. 198–201). https://doi.org/10.1659/0276-4741(2003)023{0197:VUBACA}2.0.CO;2

Linares, O. F. (1976). *``Garden hunting'' in the American tropics. Human Ecology*, *4*(4), (pp. 331–349). https://doi.org/10.1007/BF01557917

Lindström, A., & Granit, J. (2012). *Large-scale water storage in the water, energy and food nexus Perspectives on benefits, risks and best practices*. Retrieved from www.siwi.org/wp-content/uploads/2015/09/Water_Storage_Paper_21.pdf

Loh, E. H., Zambrana-Torrelio, C., Olival, K. J., Bogich, T. L., Johnson, C. K., Mazet, J. A. K., Karesh, W., & Daszak, P. (2015). Targeting Transmission Pathways for Emerging Zoonotic Disease Surveillance and Control. *Vector-Borne and Zoonotic Diseases*, *15*(7), 432–437. https://doi.org/10.1089/vbz.2013.1563

Loh, J., & Goldfinger, S. (2006). *Living Planet Report*. Gland, Switzerland.

López-Mosquera, N., & Sánchez, M. (2012). Theory of Planned Behavior and the Value-Belief-Norm Theory explaining willingness to pay for a suburban park. *Journal of Environmental Management*, *113*, (pp. 251–262). https://doi.org/10.1016/j.jenvman.2012.08.029

Lott, D. F. (2002). *American Bison: A Natural History*. Los Angeles, CAL: University of California Press.

Lu, N., & Godt, J. W. (2013). *Hillslope Hydrology and Stability*. New York, NY: Cambridge University Press.

Lugo, A. E. (2008). Visible and invisible effects of hurricanes on forest ecosystems: an international review. *Austral Ecology*, *33*(4), (pp. 368–398). https://doi.org/10.1111/j.1442-9993.2008.01894.x

Luyssaert, S., Schulze, E.-D., Börner, A., Knohl, A., Hessenmöller, D., Law, B. E., Ciais, P., & Grace, J. (2008). Old-growth forests as global carbon sinks. *Nature*, *455*(7210), 213–215. https://doi.org/10.1038/nature07276

Lyon, S. (2013). *Coffee Tourism in Chiapas: Recasting Colonial Narratives for Contemporary Markets. Culture, Agriculture, Food and Environment,* *35*(2), (pp. 125–139). https://doi.org/10.1111/cuag.12016

Lyver, P., Perez, E., Carneiro da Cunha, M., & Roué, M. (2015). *Indigenous and Local Knowledge about Pollination and Pollinators associated with Food Production: Outcomes from the Global Dialogue Workshop*. Paris.

Maass, J. M., Balvanera, P., Castillo, A., Daily, G. C., Mooney, H. A., Ehrlich, P., Quesada, M., Miranda, A., Jaramillo, V. J., & García-Oliva, F. (2005). Ecosystem services of tropical dry forests: insights from longterm ecological and social research on the Pacific Coast of Mexico. *Ecology and Society: A Journal of Integrative Science for Resilience and Sustainability*, *10*(1), 1–23.

MacDicken, K. G., Sola, P., Hall, J. E., Sabogal, C., Tadoum, M., & de Wasseige, C. (2015). Global progress toward sustainable forest management. *Forest Ecology and Management*, *352*, (pp. 47–56). https://doi.org/10.1016/j.foreco.2015.02.005

Mahon, R., Parker, C., Sinckler, T., Willoughby, S., & Johnson, J. (2007). The value of Barbados' fisheries: a preliminary assessment. *Plan Public Information Document No. 2*, 24.

Mahoney, S. P. (2009). Recreational Hunting and Sustainable Wildlife Use in North America. In B. Dickson, J. Hutton, & W. M. Adams (Eds.), *Recreational Hunting, Conservation and Rural Livelihoods* (pp. 266–281). Oxford, UK: Wiley-Blackwell. https://doi.org/10.1002/9781444303179.ch16

Marks, R. (2005). *Native Pollinators. Fish and Wildlife Habitat Management Leaflet*, (34), (pp. 1–10). https://doi.org/10.2979/NPJ.2008.9.2.80

Masters, J. (2014). *CRFM Statistics and Information Report - 2012*.

McBeth, W., Hungerford, H., Marcinkowski, T., Volk, T., & Cifranick, K. (2011). *National Environmental Literacy Assessment, Phase Two: Measuring the effectiveness of North American environmental education programs with respect to the parameters of environmental literacy*. Carbondale, IL. Retrieved from http://www.oesd.noaa.gov/pubs_reports/NELA_Phase_Two_Report_020711.pdf

McEvoy, J., & Wilder, M. (2012). Discourse and desalination: Potential impacts of proposed climate change adaptation interventions in the Arizona–Sonora border region. *Global Environmental Change*, *22*(2), (pp. 353–363). https://doi.org/10.1016/j.gloenvcha.2011.11.001

Mcivor, A., Spencer, T., & Möller, I. (2012). *Storm Surge Reduction by Mangroves* (Natural Coastal Protection No. 41). *Natural Coastal Protection Series*. Cambridge, UK. Retrieved from http://www.naturalcoastalprotection.org/documents/storm-surge-reduction-by-mangroves

McKinnon, M. C., Cheng, S. H., Dupre, S., Edmond, J., Garside, R., Glew, L., Holland, M. B., Levine, E., Masuda, Y. J., Miller, D. C., Oliveira, I., Revenaz, J., Roe, D., Shamer, S., Wilkie, D., Wongbusarakum, S., & Woodhouse, E. (2016). What are the effects of nature conservation on human well-being? A systematic map of empirical evidence from developing countries. *Environmental Evidence*, *5*(1), 8. https://doi.org/10.1186/s13750-016-0058-7

Mcleod, E., Chmura, G. L., Bouillon, S., Salm, R., Björk, M., Duarte, C. M., Lovelock, C. E., Schlesinger, W. H., & Silliman, B. R. (2011). A blueprint for blue carbon: toward an improved understanding of the role of vegetated coastal habitats in sequestering CO2. *Frontiers in Ecology and the Environment*, *9*(10), 552–560. https://doi.org/10.1890/110004

MEA. (2005). *Ecosystems and Human Well-Being: Wetlands and Water Synthesis Report*. World Resources Institute (WRI).

Meadows, D. H., Meadows, D. L., Randers, J., & III, B. W. W. (1972). *The Limits to Growth: A Report for the CUB of Rome's Project on the Predicament of Mankind* (Vol. 8). New York: Universe Books. https://doi.org/10.1111/j.1752-1688.1972.tb05230.x

Medeiros, A. S., Wood, P., Wesche, S. D., Bakaic, M., & Peters, J. F. (2017). Water security for northern peoples: review of threats to Arctic freshwater systems in Nunavut, Canada. *Regional Environmental Change*, *17*(3), (pp. 635–647). https://doi.org/10.1007/s10113-016-1084-2

Mekonnen, M. M., & Hoekstra, A. Y. (2011). *National water footprint accounts: The green, blue and grey water footprint of production and consumption* (Value of Water Research Report Series No. 50). Delft, The Netherlands.

Mekonnen, M. M., & Hoekstra, A. Y. (2012). A Global Assessment of the Water Footprint of Farm Animal Products. *Ecosystems*, *15*(3), (pp. 401–415). https://doi.org/10.1007/s10021-011-9517-8

Mendoza, G. M. (2010). *Los parientes silvestres del cultivo de la yuca en Bolivia: Estado de conocimiento, grado de conservación y acciones de conservación propuestas.* La Paz, Bolivia.

Michalak, A. M., Anderson, E. J., Beletsky, D., Boland, S., Bosch, N. S., Bridgeman, T. B., Chaffin, J. D., Cho, K., Confesor, R., Daloglu, I., Depinto, J. V, Evans, M. A., Fahnenstiel, G. L., He, L., Ho, J. C., Jenkins, L., Johengen, T. H., Kuo, K. C., Laporte, E., Liu, X., McWilliams, M. R., Moore, M. R., Posselt, D. J., Richards, R. P., Scavia, D., Steiner, A. L., Verhamme, E., Wright, D. M., & Zagorski, M. A. (2013). Record-setting algal bloom in Lake Erie caused by agricultural and meteorological trends consistent with expected future conditions. *Proceedings of the National Academy of Sciences of the United States of America*, *110*(16), 6448–6452. https://doi.org/10.1073/pnas.1216006110

Millennium Ecosystem Assessment. (2005). *Ecosystem and human well-being. Summary for decision makers. In Ecosystems and Human Well-being: Synthesis* (pp. 1–24). Washington, D.C.: Island Press. https://doi.org/http,//www.maweb.org/.

Mitchell, J. E. (2000). *Rangeland resource trends in the United States: A technical document supporting the 2000 USDA Forest Service RPA Assessment*. *USDA Forest Service General Techical Report*. Fort Collins, CO. https://doi.org/http://www.fs.fed.us/rm/pubs/rmrs%5Fgtr68.html

Mitchell, S. J. (2013). *Wind as a natural disturbance agent in forests: A synthesis*. *Forestry*, *86*(2), (pp. 147–157). https://doi.org/10.1093/forestry/cps058

Mitsch, W. J., & Gosselink, J. G. (2015). *Wetlands* (5th ed.). Hoboken, NJ, USA: Wiley.

Mitsch, W. J., Day, J. W., Gilliam, J. W., Groffman, P. M., Hey, D. L., Randall, G. W., & Wang, N. (2001). Reducing Nitrogen Loading to the Gulf of Mexico from the Mississippi River Basin: Strategies to Counter a Persistent Ecological ProblemEcotechnology—the use of natural ecosystems to solve environmental problems—should be a part of efforts to shrink the zone of hypoxia in the Gulf of Mexico. *BioScience*, *51*(5), 373–388. https://doi.org/10.1641/0006-3568(2001)051[0373:rnlttg]2.0.co;2

Mittermeier, R. A. (1991). *Hunting and its effect on wild primate populations in Suriname. In J. Robinson & K. Redford (Eds.), Neotropical wildlife use and conservation* (pp. 93–107). University of Chicago.

Moerman, D. E. (1996). An analysis of the food plants and drug plants of native North America. *Journal of Ethnopharmacology*, *52*(1), (pp. 1–22). https://doi.org/http://dx.doi.org/10.1016/0378-8741(96)01393-1

Mora, C., Graham, N. A. J., & Nyström, M. (2016). Ecological limitations to the resilience of coral reefs. *Coral Reefs*, *35*(4), (pp. 1271–1280). https://doi.org/10.1007/s00338-016-1479-z

Morales, C., & Aizen, M. A. (2004). Potential displacement of the native bumblebee Bombus dahlbomii by the invasive Bombus ruderatus in NW Patagonia. In D. de Jong (Ed.), *Proceedings of the 8th International Conference on Tropical Bees and VI Encontro sobre Abelhas* (pp. 70–76). International Bee Research Association.

Mora-Portuguez, J., & Dubois-Cisneros, V. (2015). *Implementación del derecho humano al agua en América Latina*. *VII Foro Mundial del Agua*. Buenos Aires: Corporación Andina de Fomento. Retrieved from http://files/59/CAF Implementacion del derecho humano al agua America del Sur.pdf

Morello, J., & Rodríguez, A. (2001). Parasitismo y mutualismo entre Buenos Aires y la Pampa. *Encrucijadas 10, Revista de La Universidad de Buenos Aires*.

Moreno, M. (2011). *Contributions of the Existence of National Parks and Biological Reserves in Costa Rica. In Protected Areas - not just for biodiversity Conservation: The contributions of protected areas to the economic and social development in Bhutan, Costa Rica and Benin* (pp. 251–298). UNA, CINPE, CEBEDES, NCD, Zeta Servicios Gráficos S.A.

Morton, R. A., & Barras, J. A. (2011). Hurricane Impacts on Coastal Wetlands: A Half-Century Record of Storm-Generated Features from Southern Louisiana. *Journal of Coastal Research*, *275*, (pp. 27–43). https://doi.org/10.2112/JCOASTRES-D-10-00185.1

Mouchet, M. A., Lamarque, P., Martín-López, B., Crouzat, E., Gos, P., Byczek, C., & Lavorel, S. (2014). An interdisciplinary methodological guide for quantifying associations between ecosystem services. *Global Environmental Change*, *28*(1), (pp. 298–308). https://doi.org/10.1016/j.gloenvcha.2014.07.012

Mueller, J. M., Swaffar, W., Nielsen, E. A., Springer, A. E., & Lopez, S. M. (2013). Estimating the value of watershed services following forest restoration. *Water Resources Research*, *49*(4), (pp. 1773–1781). https://doi.org/10.1002/wrcr.20163

Myers, N., Mittermeier, R. A., Mittermeier, C. G., da Fonseca, G. A. B., & Kent, J. (2000). Biodiversity hotspots for conservation priorities. *Nature*, *403*(6772), (pp. 853–858). https://doi.org/10.1038/35002501

Nagendra, H. (2008). Do Parks Work? Impact of Protected Areas on Land Cover Clearing. *AMBIO: A Journal of the Human Environment*, *37*(5), (pp. 330–337). https://doi.org/10.1579/06-R-184.1

Nahlik, A. M., & Fennessy, M. S. (2016). Carbon storage in US wetlands. *Nature Communications*, *7*, 13835. https://doi.org/10.1038/ncomms13835

Nahlik, A. M., & Mitsch, W. J. (2011). Methane emissions from tropical freshwater wetlands located in different climatic zones of Costa Rica. *Global Change Biology*, *17*(3), (pp. 1321–1334).

https://doi.org/10.1111/j.1365-2486.2010.02190.x
Nakashima, D., & Roué, M. (2002). Indigenous Knowledge , Peoples and Sustainable Practice. In *Encyclopedia of Global Environmental Change* (Timmerman, Vol. 5, pp. 314–324). Chichester: John Wiley & Sons.
Nantapo, C. W. T., Muchenje, V., Nkukwana, T. T., Hugo, A., Descalzo, A., Grigioni, G., & Hoffman, L. C. (2015). Socio-economic dynamics and innovative technologies affecting health-related lipid content in diets: Implications on global food and nutrition security. *Food Research International*, *76*, (pp. 896–905). https://doi.org/10.1016/j.foodres.2015.05.033
National Academy of Sciences. (2016). *Progress Toward Restoring the Everglades*. Washington, D.C.: National Academies Press. https://doi.org/10.17226/23672
National Marine Fisheries Service (NMFS). (2016). *Fisheries economics of the United States 2014* (No. NMFS–F/SPO-163). Washington, D.C.
National Research Council (NRC). (1996). *Upstream: Salmon and Society in the Pacific Northwest*. *Sciences-New York*. Washington, D.C.: National Academy Press.
National Research Council. (2005). *Valuing Ecosystem Services: Toward Better Environmental Decision Making*. Washington, D.C.: National Academy Press.
Nations, U. (2015). *Knowing our Lands and Resources*. *Knowing our Lands and Resources*.
NAWCC. (2017). *North American Wetland Conservation Council*.
Nelson, A., & Chomitz, K. M. (2011). Effectiveness of strict vs. multiple use protected areas in reducing tropical forest fires: A global analysis using matching methods. *PLoS ONE*, *6*(8). https://doi.org/10.1371/journal.pone.0022722
Nelson, J. L., Zavaleta, E. S., & Chapin, F. S. (2008). Boreal fire effects on subsistence resources in Alaska and adjacent Canada. *Ecosystems*, *11*(1), (pp. 156–171). https://doi.org/10.1007/s10021-007-9114-z
Nesheim, M. C., & Yaktine, A. L. (2007). *Seafood Choices: Balancing Benefits and Risks*. *The National Academies Press*. Washington, D.C. https://doi.org/10.17226/11762
Neumann, G., Noda, T., & Kawaoka, Y. (2009). Emergence and pandemic potential of swine-origin H1N1 influenza virus. *Nature*, *459*(7249), (pp. 931–939). https://doi.org/10.1038/nature08157
Niemelä, J. (2011). *Urban Ecology: Patterns, Processes and Aplications*. (J. H. Breuste, T. Elmqvist, G. Guntenspergen, P. James, & N. E. McIntyre, Eds.). Oxford University Press.
Niemelä, J., Saarela, S.-R., Söderman, T., Kopperoinen, L., Yli-Pelkonen, V., Väre, S., & Kotze, D. J. (2010). Using the ecosystem services approach for better planning and conservation of urban green spaces: a Finland case study. *Biodiversity and Conservation*, *19*(11), (pp. 3225–3243). https://doi.org/10.1007/s10531-010-9888-8
Nigam, P. S., & Singh, A. (2011). Production of liquid biofuels from renewable resources. *Progress in Energy and Combustion Science*, *37*(1), (pp. 52–68). https://doi.org/https://doi.org/10.1016/j.pecs.2010.01.003
Norman, R., Bowers, R. G., Begon, M., & Hudson, P. J. (1999). Persistence of Tick-borne Virus in the Presence of Multiple Host Species: Tick Reservoirs and Parasite Mediated Competition. *Journal of Theoretical Biology*, *200*(1), 111–118. https://doi.org/10.1006/jtbi.1999.0982
Novel Swine-Origin Influenza A (H1N1) Virus Investigation Team, DNovel Swine-Origin Influenza A (H1N1) Virus Investigation Team, Dawood FS, Jain S, Finelli L, Shaw MW, Lindstrom S, Garten RJ, Gubareva LV, Xu X, Bridges CB, Uyeki TMawood FS, Jain S, Finell, U. T. (2009). Emergence of a Novel Swine-Origin Influenza A (H1N1) Virus in Humans. *New England Journal of Medicine*, *360*(25), (pp. 2605–2615). https://doi.org/10.1056/NEJMoa0903810
O'Gara, B. W., & Yoakum, J. D. (2004). *Pronghorn: Ecology and Management*. (R. E. McCabe, Ed.). University Press of Colorado.
OECD. (2008). *Growing Unequal?: Income Distribution and Poverty in OECD Countries*. OECD Publishing.
Omi, P. N. (2005). *Forest Fires*. Santa Barbara, CA: ABC-CLIO.
ONU-HABITAT. (2012). *Estado de las ciudades de América Latina y el Caribe, 2012. Rumbo a una nueva transición urbana*. *Exit imagen y cultura*. Programa de Naciones Unidas para los Asentamientos Humanos. https://doi.org/HS/053/12S ISBN Serie 978-92-1-133397-8
Ostfeld, R. S. (2017). *Biodiversity loss and the ecology of infectious disease*. *The Lancet Planetary Health*, *1*(1), e2–e3. https://doi.org/10.1016/S2542-5196(17)30010-4

Ostfeld, R. S., & Holt, R. D. (2004). Are predators good for your health? Evaluating evidence for top-down regulation of zoonotic disease reservoirs. *Frontiers in Ecology and the Environment*, *2*(1), (pp. 13–20). https://doi.org/10.1890/1540-9295(2004)002{0013:APGFYH}2.0.CO;2

Ostfeld, R. S., & Keesing, F. (2000). Biodiversity series: The function of biodiversity in the ecology of vector-borne zoonotic diseases. *Canadian Journal of Zoology*, *78*(12), (pp. 2061–2078). https://doi.org/10.1139/cjz-78-12-2061

Ostfeld, R. S., & Keesing, F. (2012). Effects of Host Diversity on Infectious Disease. *Annual Review of Ecology, Evolution, and Systematics*, *43*(1), (pp. 157–182). https://doi.org/10.1146/annurev-ecolsys-102710-145022

Oswalt, S. N., & Smith, W. B. (2014). *US forest resource facts and historical trends.*

Overbeck, G. E., Vélez-Martin, E., Scarano, F. R., Lewinsohn, T. M., Fonseca, C. R., Meyer, S. T., Muller, S. C., Ceotto, P., Dadalt, L., Durigan, G., Ganade, G., Gossner, M. M., Guadagnin, D. L., Lorenzen, K., Jacobi, C. M., Weisser, W. W., & Pillar, V. D. (2015). Conservation in Brazil needs to include non-forest ecosystems. *Diversity and Distributions*, *21*(12), 1455–1460. https://doi.org/10.1111/ddi.12380

Paerl, H. W., & Huisman, J. (2008). CLIMATE: Blooms Like It Hot. *Science*, *320*(5872), (pp. 57–58). https://doi.org/10.1126/science.1155398

Pagiola, S., Bishop, J., & Landel-Mills, N. (2002). *Selling Forest Environmental Services: Market-Based Mechanisms for Conservation and Development*. London and Washington: Earthscan.

Palhares, R. M., Drummond, M. G., Dos Santos Alves Figueiredo Brasil, B., Cosenza, G. P., Das Graças Lins Brandão, M., & Oliveira, G. (2015). Medicinal plants recommended by the world health organization: DNA barcode identification associated with chemical analyses guarantees their quality. *PLoS ONE*, *10*(5), (pp. 1–29). https://doi.org/10.1371/journal.pone.0127866

Palmer, M. A. (2010). *Water resources: Beyond infrastructure. Nature*, *467*(7315), (pp. 534–535). Retrieved from http://dx.doi.org/10.1038/467534a

Palmer, M. A., Lettenmaier, D. P., Poff, N. L., Postel, S. L., Richter, B., & Warner, R. (2009). Climate change and river ecosystems: Protection and adaptation options. *Environmental Management*, *44*(6), (pp. 1053–1068). https://doi.org/10.1007/s00267-009-9329-1

Pascual, U., Balvanera, P., Díaz, S., Pataki, G., Roth, E., Stenseke, M., Watson, R. T., Başak Dessane, E., Islar, M., Kelemen, E., Maris, V., Quaas, M., Subramanian, S. M., Wittmer, H., Adlan, A., Ahn, S., Al-Hafedh, Y. S., Amankwah, E., Asah, S. T., Berry, P., Bilgin, A., Breslow, S. J., Bullock, C., Cáceres, D., Daly-Hassen, H., Figueroa, E., Golden, C. D., Gómez-Baggethun, E., González-Jiménez, D., Houdet, J., Keune, H., Kumar, R., Ma, K., May, P. H., Mead, A., O'Farrell, P., Pandit, R., Pengue, W., Pichis-Madruga, R., Popa, F., Preston, S., Pacheco-Balanza, D., Saarikoski, H., Strassburg, B. B., van den Belt, M., Verma, M., Wickson, F., & Yagi, N. (2017). Valuing nature's contributions to people: the IPBES approach. *Current Opinion in Environmental Sustainability*, *26–27*, 7–16. https://doi.org/10.1016/J.COSUST.2016.12.006

Pashaei Kamali, F., Meuwissen, M. P. M., de Boer, I. J. M., van Middelaar, C. E., Moreira, A., & Oude Lansink, A. G. J. M. (2016). Evaluation of the environmental, economic, and social performance of soybean farming systems in southern Brazil. *Journal of Cleaner Production*. https://doi.org/10.1016/j.jclepro.2016.03.135

Pattanayak, S. K., & Yasuoka, J. (2008). Deforestation and malaria: Revisiting the human ecology perspective. In C. J. P. Colfer (Ed.), *People, Health and Forests: A Global Interdisciplinary Overview* (pp. 197–212). https://doi.org/10.4324/9781849771627

Patz, J. A., Campbell-Lendrum, D., Holloway, T., & Foley, J. A. (2005). Impact of regional climate change on human health. *Nature*, *438*, (pp. 310). Retrieved from http://dx.doi.org/10.1038/nature04188

Pauchard, A., & Barbosa, O. (2013). Regional Assessment of Latin America: Rapid Urban Development and Social Economic Inequity Threaten Biodiversity Hotspots. In T. Elmqvist, M. Fragkias, J. Goodness, B. Güneralp, P. J. Marcotullio, R. I. McDonald, … C. Wilkinson (Eds.), *Urbanization, Biodiversity and Ecosystem Services: Challenges and Opportunities: A Global Assessment* (pp. 589–608). Dordrecht: Springer Netherlands. https://doi.org/10.1007/978-94-007-7088-1_28

Pauly, D., Christensen, V., Guénette, S., Pitcher, T. J., Sumaila, U. R., Walters, C. J., Watson, R., & Zeller, D. (2002). Towards sustainability in world fisheries. *Nature*, *418*(6898), 689–695.

https://doi.org/10.1038/nature01017

Pearce, D. W. (2001). *The Economic Value of Forest Ecosystems. Ecosystem Health, 7(4),* (pp. 284–296). https://doi.org/10.1046/j.1526-0992.2001.01037.x

Pecl, G. T., Araújo, M. B., Bell, J. D., Blanchard, J., Bonebrake, T. C., Chen, I.-C., Clark, T. D., Colwell, R. K., Danielsen, F., Evengård, B., Falconi, L., Ferrier, S., Frusher, S., Garcia, R. A., Griffis, R. B., Hobday, A. J., Janion-Scheepers, C., Jarzyna, M. A., Jennings, S., Lenoir, J., Linnetved, H. I., Martin, V. Y., McCormack, P. C., McDonald, J., Mitchell, N. J., Mustonen, T., Pandolfi, J. M., Pettorelli, N., Popova, E., Robinson, S. A., Scheffers, B. R., Shaw, J. D., Sorte, C. J. B., Strugnell, J. M., Sunday, J. M., Tuanmu, M.-N., Vergés, A., Villanueva, C., Wernberg, T., Wapstra, E., & Williams, S. E. (2017). Biodiversity redistribution under climate change: Impacts on ecosystems and human well-being. *Science (New York, N.Y.)*, *355*(6332), eaai9214. https://doi.org/10.1126/science.aai9214

Peres, C. A. (1990). *Effects of hunting on western Amazonian primate communities. Biological Conservation, 54*(1), (pp. 47–59). https://doi.org/10.1016/0006-3207(90)90041-M

Peres, C. A. (1991). *Humboldt's woolly monkeys decimated by hunting in Amazonia. Oryx*, *25*(2), (pp. 89–95). https://doi.org/10.1017/S0030605300035122

Perring, M. P., Standish, R. J., Price, J. N., Craig, M. D., Erickson, T. E., Ruthrof, K. X., Whiteley, A. S., Valentine, L. E., & Hobbs, R. J. (2015). Advances in restoration ecology: rising to the challenges of the coming decades. *Ecosphere*, *6*(8), art131. https://doi.org/10.1890/ES15-00121.1

Pershing, A. J., Alexander, M. A., Hernandez, C. M., Kerr, L. A., Le Bris, A., Mills, K. E., Nye, J. A., Record, N. R., Scannell, H. A., Scott, J. D., Sherwood, G. D., & Thomas, A. C. (2015). Slow adaptation in the face of rapid warming leads to collapse of the Gulf of Maine cod fishery. *Science (New York, N.Y.)*, *350*(6262), 809–812. https://doi.org/10.1126/science.aac9819

Peterjohn, W. T., & Correll, D. L. (1984). Nutrient Dynamics in an Agricultural Watershed: Observations on the Role of A Riparian Forest. *Ecology*, *65*(5), (pp. 1466–1475). https://doi.org/10.2307/1939127

Pichón, F. J., & Uquillas, J. E. (1997). Agricultural Intensification and Poverty Reduction in Latin America â€™ s Risk-Prone Areas : Opportunities and Challenges Published by : College of Business , Tennessee State University Stable URL : http://www.jstor.org/stable/4192714 Agricultural Intensifi. *The Journal of Developing Areas*, *31*(4), (pp. 479–514).

Pinkus-Rendon, M. A., Parra-Tabla, V., & Meléndez-Ramírez, V. (2005). Floral resource use and interactions between Apis mellifera and native bees in cucurbit crops in Yucatán, México. *The Canadian Entomologist*, *137*(4), (pp. 441–449). https://doi.org/DOI: 10.4039/n04-043

Piperno, D. R. (2011). *The Origins of Plant Cultivation and Domestication in the New World Tropics. Current Anthropology, 52*(S4), S453–S470. https://doi.org/10.1086/659998, https://www.journals.uchicago.edu/toc/ca/current.

Pirk, C. W. W., Crewe, R. M., & Moritz, R. F. A. (2017). Risks and benefits of the biological interface between managed and wild bee pollinators. *Functional Ecology*, *31*, (pp. 47–55). https://doi.org/10.1111/1365-2435.12768

Poppy, G. M., Chiotha, S., Eigenbrod, F., Harvey, C. A., Honzak, M., Hudson, M. D., Jarvis, A., Madise, N. J., Schreckenberg, K., Shackleton, C. M., Villa, F., & Dawson, T. P. (2014). Food security in a perfect storm: using the ecosystem services framework to increase understanding. *Philosophical Transactions of the Royal Society B: Biological Sciences*, *369*(1639), 20120288–20120288. https://doi.org/10.1098/rstb.2012.0288

Portillo-Quintero, C. A., & Sánchez-Azofeifa, G. A. (2010). Extent and conservation of tropical dry forests in the Americas. *Biological Conservation*, *143*(1), (pp. 144–155). https://doi.org/10.1016/j.biocon.2009.09.020

Postel, S. L. (2000). *Entering an Era of Water Scarcity : the Challenges Ahead. Ecological Applications*, *10*(4), (pp. 941–948). https://doi.org/10.1890/1051-0761(2000)010{0941:EAEOWS}2.0.CO;2

Postel, S. L., & Thompson, B. H. (2005). Watershed protection: Capturing the benefits of nature's water supply services. *Natural Resources Forum*, *29*(2), (pp. 98–108). https://doi.org/10.1111/j.1477-8947.2005.00119.x

Potts, S. G., Biesmeijer, J. C., Kremen, C., Neumann, P., Schweiger, O., & Kunin, W. E. (2010). Global pollinator declines: Trends, impacts and drivers. *Trends in Ecology and Evolution*, *25*(6), (pp. 345–353). https://doi.org/10.1016/j.tree.2010.01.007

Pou, S. A., Díaz, P., Gabriela, A., Quintana, D. La, Forte, C. A., & Aballay, L. R. (2016). Identification of dietary patterns in urban population of Argentina : study on diet-obesity relation in population-based prevalence study. *Nutrition Research and Practice*, *10*(6), (pp. 616–622). https://doi.org/10.4162/nrp.2016.10.6.616

Power, A. G. (2010). *Ecosystem services and agriculture: tradeoffs and synergies. Philosophical Transactions of the Royal Society of London. Series B, Biological Sciences*, *365*(1554), (pp. 2959–2971). https://doi.org/10.1098/rstb.2010.0143

Power, E. M. (2008). *Conceptualizing food security for aboriginal people in Canada. Canadian Journal of Public Health*, *99*(2), (pp. 95–97). https://doi.org/10.1016/j.gloenvcha.2007.09.002

Pretty, J. (2008). *Agricultural sustainability: concepts, principles and evidence. Philosophical Transactions of the Royal Society of London. Series B, Biological Sciences*, *363*(1491), (pp. 447–65). https://doi.org/10.1098/rstb.2007.2163

Pretty, J., Sutherland, W. J., Ashby, J., Auburn, J., Baulcombe, D., Bell, M., Bentley, J., Bickersteth, S., Brown, K., Burke, J., Campbell, H., Chen, K., Crowley, E., Crute, I., Dobbelaere, D., Edwards-Jones, G., Funes-Monzote, F., Godfray, H. C. J., Griffon, M., Gypmantisiri, P., Haddad, L., Halavatau, S., Herren, H., Holderness, M., Izac, A.-M., Jones, M., Koohafkan, P., Lal, R., Lang, T., McNeely, J., Mueller, A., Nisbett, N., Noble, A., Pingali, P., Pinto, Y., Rabbinge, R., Ravindranath, N. H., Rola, A., Roling, N., Sage, C., Settle, W., Sha, J. M., Shiming, L., Simons, T., Smith, P., Strzepeck, K., Swaine, H., Terry, E., Tomich, T. P., Toulmin, C., Trigo, E., Twomlow, S., Vis, J. K., Wilson, J., & Pilgrim, S. (2010). The top 100 questions of importance to the future of global agriculture. *International Journal of Agricultural Sustainability*, *8*(4), 219–236. https://doi.org/10.3763/ijas.2010.0534

Pullin, A. S., Bangpan, M., Dalrymple, S., Dickson, K., Haddaway, N. R., Healey, J. R., Hauari, H., Hockley, N., Jones, J. P. G., Knight, T., Vigurs, C., & Oliver, S. (2013). Human well-being impacts of terrestrial protected areas. *Environmental Evidence*, *2*(1), 19. https://doi.org/10.1186/2047-2382-2-19

Purkey, D. R., Thomas, G. A., Fullerton, D. K., Moench, M., & Axelrad, L. (1998). *Feasibility study of a maximal program of groundwater banking*. Berkeley: Natural Heritage Institute.

Rader, R., Bartomeus, I., Garibaldi, L. A., Garratt, M. P. D., Howlett, B. G., Winfree, R., … Woyciechowski, M. (2015). Non-bee insects are important contributors to global crop pollination. *Proceedings of the National Academy of Sciences*, 201517092. https://doi.org/10.1073/pnas.1517092112

Ranum, P., Peña-Rosas, J. P., & Garcia-Casal, M. N. (2014). Global maize production, utilization, and consumption. *Annals of the New York Academy of Sciences*, *1312*(1), (pp. 105–112). https://doi.org/10.1111/nyas.12396

Rassy, D., & Smith, R. D. (2013). The economic impact of H1N1 on Mexico’s tourist and pork sectors. *Health Economics*, *22*(7), (pp. 824–834). https://doi.org/10.1002/hec.2862

Raudsepp-Hearne, C., Peterson, G. D., & Bennett, E. M. (2010). Ecosystem service bundles for analyzing tradeoffs in diverse landscapes. *Proceedings of the National Academy of Sciences of the United States of America*, *107*(11), (pp. 5242–7). https://doi.org/10.1073/pnas.0907284107

Raven, J., Caldeira, K., Elderfield, H., Hoegh-Guldberg, O., Liss, P., Riebesell, U., Shepherd, J., Turley, C., & Watson, A. (2005). *Ocean acidification due to increasing atmospheric carbon dioxide*. *Policy document 12/05*. Cardiff, UK. https://doi.org/ISBN 0 85403 617 2

Reichel-Dolmatoff, G. (1971). *Amazonian Cosmos: The Sexual and Religious Symbolism of the Tukano Indians*. University of Chicago Press.

Ren, D. (2015). *Storm-triggered Landslides in Warmer Climates*. New York, NY: Springer International Publishing Switzerland.

Ricketts, T. H., Daily, G. C., Ehrlich, P. R., & Michener, C. D. (2004). Economic value of tropical forest to coffee production. *Proceedings of the National Academy of Sciences of the United States of America*, *101*(34), (pp. 12579–82). https://doi.org/10.1073/pnas.0405147101

Rittl, T. F., Oliveira, D., & Cerri, C. E. P. (2017). Soil carbon stock changes under different land uses in the Amazon. *Geoderma Regional*, *10*, (pp. 138–143). https://doi.org/10.1016/j.geodrs.2017.07.004

Riviére, G. (1994). *El sistema de aynuqa: Memoria e historia de la comunidad (comunidades aymara del altiplano boliviano). In D. Hervé, D. Genin, & G. Rivière (Eds.), Dinámicas del descanso de la tierra en los Andes* (pp. 89–105). La Paz, Bolivia: IBTA - ORSTOM.

Rizo-Decelis, L. D., & Andreo, B. (2016). Water Quality Assessment of the Santiago River and Attenuation

Capacity of Pollutants Downstream Guadalajara City, Mexico. *River Research and Applications*, *32*(7), (pp. 1505–1516). https://doi.org/10.1002/rra.2988

Roberts, L. A., & Leitch, J. A. (1997). *Economic valuation of some wetland outputs of Mud Lake, Minnesota-South Dakota*. North Dakota State University Department of Agribusiness and Applied Economics.

Robertson, G. P., Gross, K. L., Hamilton, S. K., Landis, D. A., Schmidt, T. M., Snapp, S. S., & Swinton, S. M. (2014). Farming for ecosystem services: An ecological approach to production agriculture. *BioScience*, *64*(5), (pp. 404–415). https://doi.org/10.1093/biosci/biu037

Robillard, C. M., Coristine, L. E., Soares, R. N., & Kerr, J. T. (2015). Facilitating climate-change-induced range shifts across continental land-use barriers. *Conservation Biology*, *29*(6), (pp. 1586–1595). https://doi.org/10.1111/cobi.12556

Robinson, J., & Bennett, E. (2000). *Hunting for Sustainability in Tropical Forests*. New York: Columbia University Press.

Robinson, J., & Redford, K. H. (1991). *Neotropical wildlife use and conservation*. Chicago: University of Chicago Press.

Rocha, F. L., Roque, A. L. R., Arrais, R. C., Santos, J. P., Lima, V. D. S., Xavier, S. C. D. C., Cordeir-Estrela, P., D'andrea, P. S., & Jansen, A. M. (2013). Trypanosoma cruzi TcI and TcII transmission among wild carnivores, small mammals and dogs in a conservation unit and surrounding areas, Brazil. *Parasitology*, *140*(02), 160–170. https://doi.org/10.1017/S0031182012001539

Rocha, G. O. D. A., Anjos, J. P. D. O. S., & Andrade, J. B. D. E. (2015). Energy trends and the water-energy binomium for Brazil. *Anais Da Academia Brasileira de Ciências*, *87*, 569–594. https://doi.org/10.1590/0001-3765201520140560

Rodriguez, L. (2010). *Origen y evolución de la papa cultivada. Una revisión. Agronomía Colombiana*, *28*(1), (pp. 9–17).

Rolston, H. (1986). *Philosophy Gone Wild: Essays in Environmental Ethics*. Amherst, NY: Prometheus Books.

Romanelli, C., Cooper, D., Campbell-Lendrum, D., Maiero, M., Karesh, W., Hunter, D., & Golden, C. (2015). *Connecting global priorities: biodiversity and human health: a state of knowledge review*.

Romijn, E., Lantican, C. B., Herold, M., Lindquist, E., Ochieng, R., Wijaya, A., Murdiyarso, D., & Verchot, L. (2015). Assessing change in national forest monitoring capacities of 99 tropical countries. *Forest Ecology and Management*, *352*, 109–123.

Rose, G. a., & Rowe, S. (2015). *Northern cod comeback. Canadian Journal of Fisheries and Aquatic Sciences*, *72*(October), (pp. 1789–1798). https://doi.org/10.1139/cjfas-2015-0346

Roy, S., Byrne, J., & Pickering, C. (2012). A systematic quantitative review of urban tree benefits, costs, and assessment methods across cities in different climatic zones. *Urban Forestry and Urban Greening*, *11*(4), (pp. 351–363). https://doi.org/10.1016/j.ufug.2012.06.006

Royal Botanic Gardens Kew. (2017). *State of the World's Plants 2017*. *Royal Botanic Gardens, Kew*. https://doi.org/978-1-84246-628-5

Rozzi, R., Armesto, J. J., Gutiérrez, J. R., Massardo, F., Likens, G. E., Anderson, C. B., Poole, A., Moses, K. P., Hargrove, E., Mansilla, A. O., Kennedy, J. H., Willson, M., Jax, K., Jones, C. G., Callicott, J. B., & Arroyo, M. T. K. (2012). Integrating Ecology and Environmental Ethics: Earth Stewardship in the Southern End of the Americas. *BioScience*, *62*(3), 226–236. https://doi.org/10.1525/bio.2012.62.3.4

Rozzi, R., Massardo, F., Anderson, C. B., Heidinger, K., & Silander, J. A. (2006). Ten principles for biocultural conservation at the southern tip of the Americas: The approach of the Omora Ethnobotanical Park. *Ecology and Society*, *11*(1). https://doi.org/43

Ruiz-Pérez, M., Belcher, B., Achdiawan, R., Alexiades, M., Aubertin, C., Caballero, J., Campbell, B., Clement, C., Cunningham, T., Fantini, A., de Foresta, H., García Fernández, C., Gautam, K., Hersch Martínez, P., de Jong, W., Kusters, K., Kutty, M. G., López, C., Fu, M., Alfaro, M., Angel, M., Nair, T. K. R., Ndoye, O., Ocampo, R., Rai, N., Ricker, M., Schreckenberg, K., Shackleton, S., Shanley, P., Sunderland, T., & Youn, Y.-C. (2004). Markets Drive the Specialization Strategies of Forest Peoples. *Ecology and Society*, *9*(2). https://doi.org/10.5751/ES-00655-090204

Russell, R., Guerry, A. D., Balvanera, P., Gould, R. K., Basurto, X., Chan, K. M. A., Klain, S., Levine, J., & Tam, J. (2013). Humans and Nature: How Knowing and Experiencing Nature Affect Well-Being. *Annu. Rev. Environ. Resour*, *38*, (pp. 473–502). https://doi.org/10.1146/annurev-environ-012312-

110838
Salim, A., Pierce Colfer, J. C., & McDougall, C. (1999). *Scoring and Analysis Guide for Assessing Human Well-Being*.
Salinas, E. (2010). Valor cultural de los mamíferos en Bolivia. In R. Wallace, H. Gomez, Z. Porcel, & D. Rumiz (Eds.), *Distribución, ecologia y conservación de los mamíferos medianos y grandes de Bolivia* (pp. 5–51). Santa Cruz, Bolivia: Fundación Simón Patiño.
Salkeld, D. J., Padgett, K. A., & Jones, J. H. (2013). A meta-analysis suggesting that the relationship between biodiversity and risk of zoonotic pathogen transmission is idiosyncratic. *Ecology Letters*, *16*(5), (pp. 679–686). https://doi.org/10.1111/ele.12101
Salmond, J. A., Tadaki, M., Vardoulakis, S., Arbuthnott, K., Coutts, A., Demuzere, M., Dirks, K. N., Heaviside, C., Lim, S., Macintyre, H., McInnes, R. N., & Wheeler, B. W. (2016). Health and climate related ecosystem services provided by street trees in the urban environment. *Environmental Health*, *15*(S1), S36. https://doi.org/10.1186/s12940-016-0103-6
Sánchez-Rojas, O., Medina, K. P., Gutiérrez, C., García, L. E., Marchesini, R., Prifer, K., & Ballestero, I. (2013). *Comportamiento del comercio bananero mundial- Cuaderno de la CEPAL*. San José, Costa Rica.
Sans, P., & Combris, P. (2015). World meat consumption patterns: An overview of the last fifty years (1961-2011). *Meat Science*, *109*, (pp. 106–11). https://doi.org/10.1016/j.meatsci.2015.05.012
Santos-Fita, D., Naranjo, E. J., & Rangel-Salazar, J. L. (2012). Wildlife uses and hunting patterns in rural communities of the Yucatan Peninsula, Mexico. *Journal of Ethnobiology and Ethnomedicine*, *8*(1), (pp. 38). https://doi.org/10.1186/1746-4269-8-38
Santos-Fita, D., Naranjo, E. J., Estrada, E. I. J., Mariaca, R., & Bello, E. (2015). Symbolism and ritual practices related to hunting in Maya communities from central Quintana Roo, Mexico. *Journal of Ethnobiology and Ethnomedicine*, *11*(1), (pp. 71). https://doi.org/10.1186/s13002-015-0055-x
Scharlemann, J. P., Tanner, E. V., Hiederer, R., & Kapos, V. (2014). Global soil carbon: understanding and managing the largest terrestrial carbon pool. *Carbon Management*, *5*(1), (pp. 81–91). https://doi.org/10.4155/cmt.13.77
Scherr, S. J., & McNeely, J. A. (2008). Biodiversity conservation and agricultural sustainability: towards a new paradigm of “ecoagriculture”landscapes. *Philosophical Transactions of the Royal Society B: Biological Sciences*, *363*(July 2007), (pp. 477–494). https://doi.org/10.1098/rstb.2007.2165
Scheuhammer, A., Chan, H. M., Frouin, H., Krey, A., Letcher, R., Loseto, L., Noël, M., Ostertag, S., Ross, P., & Wayland, M. (2015). Recent progress on our understanding of the biological effects of mercury in fish and wildlife in the Canadian Arctic. *Science of The Total Environment*, *509–510*, 91–103. https://doi.org/10.1016/J.SCITOTENV.2014.05.142
Schlesinger, W. H., & Bernhardt, E. S. (2013). *Biogeochemistry: An analysis of global change*. Waltham, MA: Elsevier.
Schmidt, M. J., Rapp Py-Daniel, A., de Paula Moraes, C., Valle, R. B. M., Caromano, C. F., Texeira, W. G., Barbosa, C. A., Fonseca, J. A., Magalhães, M. P., Silva do Carmo Santos, D., da Silva e Silva, R., Guapindaia, V. L., Moraes, B., Lima, H. P., Neves, E. G., & Heckenberger, M. J. (2014). Dark earths and the human built landscape in Amazonia: a widespread pattern of anthrosol formation. *Journal of Archaeological Science*, *42*, 152–165. https://doi.org/10.1016/J.JAS.2013.11.002
Schmitz, C., van Meijl, H., Kyle, P., Nelson, G. C., Fujimori, S., Gurgel, A., Havlik, P., Heyhoe, E., d’Croz, D. M., Popp, A., Sands, R., Tabeau, A., van der Mensbrugghe, D., von Lampe, M., Wise, M., Blanc, E., Hasegawa, T., Kavallari, A., & Valin, H. (2014). Land-use change trajectories up to 2050: insights from a global agro-economic model comparison. *Agricultural Economics*, *45*(1), 69–84. https://doi.org/10.1111/agec.12090
Scholte, S., van Teeffelen, A., & Verburg, P. (2015). *Integrating socio-cultural perspectives into ecosystem service valuation: A review of concepts and methods*. *Ecological Economics* (Vol. 114). https://doi.org/10.1016/j.ecolecon.2015.03.007
Schuur, E. A. G., & Bockheim, J. (2008). Vulnerability of permafrost carbon to climate change: Implications for the global carbon cycle. *BioScience*, *58*(September), (pp. 701–714). https://doi.org/10.1641/B580807
Secretariat of the Convention on Biological Diversity(SCBD). (2012). *Report of the work of the expert group on maintaining the ability of Biodiversity to continue to support the water cycle*. Retrieved from

http://www.cbd.int/doc/meetings/cop/cop-11/information/cop-11-inf-02-en.pdf

Secretariat of the Convention on Biological Diversity. (2008). *Protected areas in todays' world: Their values and benefits for the welfare of the planet*. *Diversity* (CBD Techni, Vol. 43). Convention on Biological Diversity-UNEP. Retrieved from http://citeseerx.ist.psu.edu/viewdoc/download?doi=10.1.1.165.3846&rep=rep1&type=pdf#page=10

Seidl, A. F., & Moraes, A. S. (2000). Global valuation of ecosystem services: application to the Pantanal da Nhecolandia, Brazil. *Ecological Economics*, *33*(1), (pp. 1–6). https://doi.org/https://doi.org/10.1016/S0921-8009(99)00146-9

Seppelt, R., Dormann, C. F., Eppink, F. V., Lautenbach, S., & Schmidt, S. (2011). A quantitative review of ecosystem service studies: Approaches, shortcomings and the road ahead. *Journal of Applied Ecology*, *48*(3), (pp. 630–636). https://doi.org/10.1111/j.1365-2664.2010.01952.x

Serratos Hernandez, J. A. (2009). *El origen y la diversidad del maiz en el continente americano*. *Greenpeace Mexico*. Mexico: Greenpeace Mexico.

Shanahan, D. F., Fuller, R. A., Bush, R., Lin, B. B., & Gaston, K. J. (2015). The health benefits of urban nature: How much do we need? *BioScience*, *65*(5), (pp. 476–485). https://doi.org/10.1093/biosci/biv032

Shanley, P., & Luz, L. (2003). The impacts of forest degradation on medicinal plant use and implications for health care in eastern Amazonia. *Bioscience*, *53*(6), (pp. 573–584). https://doi.org/10.1641/0006-3568(2003)053{0573:TIOFDO}2.0.CO;2

Sheoran, A. S., & Sheoran, V. (2006). Heavy metal removal mechanism of acid mine drainage in wetlands: A critical review. *Minerals Engineering*, *19*(2), (pp. 105–116). https://doi.org/10.1016/j.mineng.2005.08.006

Shi, P., & Kasperson, R. (2015). *World Atlas of Natural Disaster Risk*. (P. Shi & R. Kasperson, Eds.). New York: Springer-Verlag.

Shiklomanov, I. A., & Rodda, J. C. (2003). *World Water Resources at the Beginning of the Twenty-First Century*. Cambridge, UK: Cambridge University Press.

SIB Magallanes. (2017). *Conservación de la naturaleza: Región de Magallanes y Antártica Chilena.* Retrieved April 25, 2017, from http://sibmagallanes.cl/esp/intro-conservacion.php

Silva, R. A., Lapola, D. M., Patricio, G. B., Teixeira, M. C., Pinho, P., & Priess, J. A. (2016). Operationalizing payments for ecosystem services in Brazil's sugarcane belt: How do stakeholder opinions match with successful cases in Latin America? *Ecosystem Services*, *22*, *Part A*, (pp. 128–138). https://doi.org/http://doi.org/10.1016/j.ecoser.2016.09.013

Silvius, K. M., Bodmer, R. E., & Fragoso, J. M. V. (2004). *People in Nature: Wildlife Conservation in South and Central America*. Columbia University Press.

Simons, G. F., & Fennig, C. D. (2017). *Ethnologue: Languages of the World* (Twentieth). Dallas, TX: SIL International.

Singh, M. D., Morris, M. J., Guimarães, D. A., Bourne, G., & Garcia, G. W. (2016). Serological evaluation of ovarian steroids of red-rumped agouti (Dasyprocta leporina) during the estrous cycle phases. *Animal Reproduction Science*, *175*, (pp. 27–32). https://doi.org/http://doi.org/10.1016/j.anireprosci.2016.10.005

Sippo, J. Z., Maher, D. T., Tait, D. R., Holloway, C., & Santos, I. R. (2016). Are mangroves drivers or buffers of coastal acidification? Insights from alkalinity and dissolved inorganic carbon export estimates across a latitudinal transect. *Global Biogeochemical Cycles*, *30*(5), (pp. 753–766). https://doi.org/10.1002/2015GB005324

Siraj, A. S., Santos-Vega, M., Bouma, M. J., Yadeta, D., Carrascal, D. R., & Pascual, M. (2014). Altitudinal Changes in Malaria Incidence in Highlands of Ethiopia and Colombia. *Science*, *343*(6175), 1154 LP-1158. Retrieved from http://science.sciencemag.org/content/343/6175/1154.abstract

Smith, K. (2013). *Environmental Hazards: Assessing Risk and Reducing Disaster*. Routledge.

Smith, L. M., Case, J. L., Smith, H. M., Harwell, L. C., & Summers, J. K. (2013). Relating ecoystem services to domains of human well-being: Foundation for a U.S. index. *Ecological Indicators*, *28*, (pp. 79–90). https://doi.org/10.1016/j.ecolind.2012.02.032

Smith, P., Ashmore, M. R., Black, H. I. J., Burgess, P. J., Evans, C. D., Quine, T. A., Thomson, A. M., Hicks, K., & Orr, H. G. (2013). REVIEW: The role of ecosystems and their management in regulating climate, and soil, water and air quality. *Journal of Applied Ecology*, *50*(4), 812–829. https://doi.org/10.1111/1365-2664.12016

Smith-Ramírez, C., Ramos-Jiliberto, R., Valdovinos, F. S., Martínez, P., Castillo, J. A., & Armesto, J. J. (2014). Decadal trends in the pollinator assemblage of Eucryphia cordifolia in Chilean rainforests. *Oecologia*, *176*(1), 157–169. https://doi.org/10.1007/s00442-014-3000-0

Smyth, C. E., Stinson, G., Neilson, E., Lemprière, T. C., Hafer, M., Rampley, G. J., & Kurz, W. A. (2014). Quantifying the biophysical climate change mitigation potential of Canada's forest sector. *Biogeosciences*, *11*(13), (pp. 3515–3529). https://doi.org/10.5194/bg-11-3515-2014

Soares-Filho, B. S., Nepstad, D. C., Curran, L. M., Cerqueira, G. C., Garcia, R. A., Ramos, C. A., Voll, E., McDonald, A., Lefebvre, P., & Schlesinger, P. (2006). Modelling conservation in the Amazon basin. Nature, 440(7083), 520–523. https://doi.org/10.1038/nature04389

Soga, M., & Gaston, K. J. (2016). Extinction of experience: the loss of human–nature interactions. *Frontiers in Ecology and the Environment*, *14*(2), (pp. 94–101). https://doi.org/10.1002/fee.1225

Soloman, S. (2010). *Water: The Epic Struggle of Wealth, Power and Civilization*. New York, NY: HarperCollins Publishers.

Sousa e Silva, G. de, Costa, E., Bernardo, F. A., Sauter Groff, F. H., Todeschini, B., Viali dos Santos, D., & Machado, G. (2014). Panorama da bovinocultura no Rio Grande do Sul. *Acta Scientiae Veterinariae*, *42*(199), (pp. 1–7). https://doi.org/1215

Soutullo, A., & Gudynas, E. (2006). How effective is the MERCOSUR's network of protected areas in representing South America's ecoregions? *Oryx*, *40*(1), (pp. 112–116). https://doi.org/DOI: 10.1017/S0030605306000020

Spalding, M., Mclvor, A., Tonneijck, F., Tol, S., & van Eijk, P. (2014). *Mangroves for coastal defence. Guidelines for coastal managers & policy makers*.

Stallman, H. R. (2011). Ecosystem services in agriculture: Determining suitability for provision by collective management. *Ecological Economics*, *71*, (pp. 131–139). https://doi.org/10.1016/j.ecolecon.2011.08.016

Starkey, P. (2010). *Livestock for traction: world trends, key issues and policy implications*.

Starr, G. C., Lal, R., Malone, R., Hothem, D., Owens, L., & Kimble, J. (2000). Modeling soil carbon transported by water erosion processes. *Land Degradation & Development*, *11*(1), (pp. 83–91). https://doi.org/10.1002/(SICI)1099-145X(200001/02)11:1<83::AID-LDR370>3.0.CO;2-W

Steel, Z. L., Steel, A. E., Williams, J. N., Viers, J. H., Marquet, P. A., & Barbosa, O. (2017). Patterns of bird diversity and habitat use in mixed vineyard-matorral landscapes of Central Chile. *Ecological Indicators*, *73*, (pp. 345–357). https://doi.org/http://doi.org/10.1016/j.ecolind.2016.09.039

Steg, L., van den Berg, A. E., & de Groot, J. I. M. (2012). *Environmental Psychology: An Introduction*. BPS Blackwell.

Sun, S., Sun, G., Caldwell, P., McNulty, S., Cohen, E., Xiao, J., & Zhang, Y. (2015). Drought impacts on ecosystem functions of the U.S. National Forests and Grasslands: Part II assessment results and management implications. *Forest Ecology and Management*, *353*, (pp. 269–279). https://doi.org/10.1016/j.foreco.2015.04.002

Sutherland, W. J. (2003). *Parallel extinction risk and global distribution of languages and species. Nature, 423(6937)*, (pp. 276–279). Retrieved from http://dx.doi.org/10.1038/nature01607

Suzán, G., Marcé, E., Giermakowski, J. T., Mills, J. N., Ceballos, G., Ostfeld, R. S., Armién, B., Pascale, J. M., & Yates, T. L. (2009). Experimental Evidence for Reduced Rodent Diversity Causing Increased Hantavirus Prevalence. *PLoS ONE*, *4*(5), e5461. https://doi.org/10.1371/journal.pone.0005461

Tanaka, J. a, Brunson, M., & Torell, L. A. (2011). A Social and Economic Assessment of Rangeland Conservation Practices. In *Conservation Benefits of Rangeland Practices: Assessment, Recommendations, and Knowledge Gaps* (pp. 371–422).

Tarnocai, C. (2006). The effect of climate change on carbon in Canadian peatlands. *Global and Planetary Change*, *53*(4), (pp. 222–232). https://doi.org/10.1016/j.gloplacha.2006.03.012

Taylor, L. H., Latham, S. M., & Woolhouse, M. E. J. (2001). Risk factors for human disease emergence. *Philosophical Transactions of the Royal Society B: Biological Sciences*, *356*(1411), (pp. 983–989). https://doi.org/10.1098/rstb.2001.0888

Tehel, A., Brown, M. J. F., & Paxton, R. J. (2016). Impact of managed honey bee viruses on wild bees. *Current*

Opinion in Virology, *19*, (pp. 16–22). https://doi.org/10.1016/j.coviro.2016.06.006

Temper, L., Bene, D. del, & Martinez-Alier, J. (2015). Mapping the frontiers and frontlines of global e nvironmental justice: the EJAtlas. *Journal of Political Ecology*, *22*(266642), (pp. 255–278).

Terborgh, J., Lopez, L., Nuñez, P., Rao, M., Shahabuddin, G., Orihuela, G., Riveros, M., Ascanio, R., Adler, G. H., Lambert, T. D., & Balbas, L. (2001). Ecological meltdown in predator-free forest fragments. *Science (New York, N.Y.)*, *294*(5548), 1923–1926. https://doi.org/10.1126/science.1064397

Terrazas, F., Cadima, X., & García, R. (2008). *Catálogo etnobotánico de papas nativas: Tradición y cultura de los ayllus del norte potosí y Oruro*. Cochabamba, Bolivia: Impr. Poligraf/Fundación PROINPA.

The Mountain Institute. (1998). *Community Based Mountain Tourism, Practices for Linking Conservation with Enterprise. In Synthesis of a Mountain Forum Electronic Conference in Support of the Mountain Agenda*. Washington DC.

The World Bank. (2009). *Minding the Stock: Bringing Public Policy to Bear on Livestock Sector Development* (No. 44010–GLB). *Report N° 44010-GLB*. Washington, D.C. Retrieved from http://siteresources.worldbank.org/INTARD/Resources/FinalMindingtheStock.pdf

The World Bank. (2012). *Global Economic Prospects: Uncertainties and vulnerabilities*. *Global Economic Prospects* (Vol. 4). https://doi.org/10.1596/978-0-8213-7365-1

The World Bank. (2017). *Improved water source, urban and rural.* Retrieved from https://data.worldbank.org/data-catalog/world-development-indicators

The World Bank. (2017a). *World Bank Database*. Retrieved November 15, 2017, from https://data.worldbank.org

The World Energy Council. (2017). *World Energy: Trilemma Index 2017*. London, UK. Retrieved from https://www.worldenergy.org/wp-content/uploads/2016/10/Full-report_Energy-Trilemma-Index-2016.pdf

Thibodeau, F. R., & Ostro, B. D. (1981). An Economic Analysis of Wetland Protection. *Journal of Environmental Management*, *12*, (pp. 19–30).

Thiele-Bruhn, S., Bloem, J., de Vries, F. T., Kalbitz, K., & Wagg, C. (2012). Linking soil biodiversity and agricultural soil management. *Current Opinion in Environmental Sustainability*, *4*(5), (pp. 523–528). https://doi.org/10.1016/j.cosust.2012.06.004

Thomas, J. W., & Toweill, D. E. (1982). *Elk of North America, Ecology and Management*. Wildlife Management Institute.

Thornton, P. K. (2010). Livestock production: recent trends, future prospects. *Philosophical Transactions of the Royal Society of London.Series B, Biological Sciences*, *365*(1554), (pp. 2853–2867). https://doi.org/10.1098/rstb.2010.0134 {doi}

Tietenberg, T., & Lewis, L. (2014). *Environmental and Natural Resource Economics* (8th Editio). New York, NY: Routledge.

Toledo, V. M., & Barrera-Bassols, N. (2008). *La memoria biocultural: La importancia ecológica de las sabidurías tradicionales*.

Tolmasquim, T. M. (2016). *Energia Renovável - Hidráulica, Biomassa, Eólica, Solar, Oceânica*. *Journal of Chemical Information and Modeling* (Vol. 53). Rio de Janeiro, Brazil: Empresa de Pesquisa Energética (EPE). https://doi.org/10.1017/CBO9781107415324.004

Townsend, W. E. (2010). *Plan de Gestión Territorial Indígena TCO Baure*. Santa Cruz, Bolivia.

Townsend, W. R. (1995). *Living on the Edge: Sirionó Hunting and Fishing in Lowland Bolivia*. University of Florida.

Townsend, W. R., & Gómez, H. (2010). Roles económicos de los Mamíferos medianos y grandes de Bolivia. In R. B. Wallace, H. Gomez, Z. R. Porcel, & D. I. Rúmiz (Eds.), *Distribución, Ecología y Conservación de los Mamíferos Medianos y Grandes de Bolivia* (pp. 75–90). Santa Cruz, Bolivia: Fundación Simon Patiño.

Townsend, W. R., & Macuritofe Ramírez, V. (1995). *Cultural Teachings as an Ecological Data Base: Murui (Witoto) Knowledge About Primates*. *Latinamericanist* (Vol. 31). University of Florida. Center for Latina American Studies.

Townsend, W. R., & Rúmiz, D. I. (2003). La importancia de la fauna silvestre para las comunidades indígenas de las tierras bajas de Bolivia. In P. I. Ibisch & G. Merida (Eds.), *Biodiversidad: La riqueza de Bolivia: Estado de conocimiento y conservación.* (pp. 307–310). Santa Cruz, Bolivia: Editorial FAN.

Townsend, W. R., & Rumiz, D. I. (2004). Reflexiones sobre la posibilidad de manejo de fauna silvestre en la tierras bajas de Bolivia: experiencias comunitarias. *Revista Boliviana de Ecología Y Conservación Ambiental*, (16), (pp. 61–72). Retrieved from http://biblat.unam.mx/es/revista/revista-boliviana-de-ecologia-y-conservacion-ambiental/articulo/reflexiones-sobre-la-posibilidad-de-manejo-de-fauna-silvestre-en-la-tierras-bajas-de-bolivia-experiencias-comunitarias

Travis, C. C., & Hester, S. T. (1991). Global chemical pollution. *Environmental Science & Technology*, *25*(5), (pp. 814–819). https://doi.org/10.1021/es00017a001

Trilleras, J. M., Jaramillo, V. J., Vega, E. V., & Balvanera, P. (2015). Effects of livestock management on the supply of ecosystem services in pastures in a tropical dry region of western Mexico. *Agriculture, Ecosystems & Environment*, *211*, (pp. 133–144). https://doi.org/10.1016/j.agee.2015.06.011

Trimble, M., & Johnson, D. (2013). Artisanal fishing as an undesirable way of life? The implications for governance of fishers' wellbeing aspirations in coastal Uruguay and southeastern Brazil. *Marine Policy*, *37*, (pp. 37–44). https://doi.org/https://doi.org/10.1016/j.marpol.2012.04.002

Troy, A., & Wilson, M. A. (2006). Mapping ecosystem services: Practical challenges and opportunities in linking GIS and value transfer. *Ecological Economics*, *60*(2), (pp. 435–449). https://doi.org/https://doi.org/10.1016/j.ecolecon.2006.04.007

Tucker Lima, J. M., Valle, D., Moretto, E. M., Pulice, S. M. P., Zuca, N. L., Roquetti, D. R., Beduschi, L. E. C., Praia, A. S., Okamoto, C. P. F., da Silva Carvalhaes, V. L., Branco, E. A., Barbezani, B., Labandera, E., Timpe, K., & Kaplan, D. (2016). A social-ecological database to advance research on infrastructure development impacts in the Brazilian Amazon. *Scientific Data*, *3*, 160071. https://doi.org/10.1038/sdata.2016.71

U. S. Bureau of the Census (USBC). (2016). *Revisions to the 2014 Census of Fatal Occupational Injuries*. Washington, D.C.: U. S Department of Labor.

U.S. Fish and Wildlife Service. (2011). *National Survey of Fishing, Hunting, and Wildlife-Associated Recreation*. 172pp. Washington DC.

Ugolini, F. C., & Spaltenstein, H. (1992). Pedosphere. In S. S. Butcher, R. J. Charlson, G. H. Orians, & G. V. Wolfe (Eds.), *Global Biogeochemical Cycles* (pp. 123–153). London: Academic Press.

Uhart, M., Pérez, A. A., Rostal, M., Robles, E. A., Mendoza, A. P., Nava, A., De Paula, C. D., Miranda, F., Iñiguez, V., Zambrana, C., Durigon, E., Franco, P., Joly, D., Goldstein, T., Karesh, W. & Maret, J. (2013). A "One Health" Approach to Predict Emerging Zoonoses in the Amazon. In M. Chame & N. Labarthe (Eds.), *Wildlife and Human Health: Experiences and Perspectives* (pp. 65–73). Rio de Janeiro, Brasil: FIOCRUZ.

UNISDR/OHRLLS. (2013). *Addressing Risk, Harnessing Opportunity: Building Disaster Resilience in SIDS*. *Issue Paper*. Geneva Switzerland.

United Nations Development Program (UNDP). (2016). *Sustainable Development Goals*. Retrieved from http://www.undp.org/content/undp/en/home/sustainable-development-goals.html

United Nations Development Programme (UNDP). (2016). *Regional Human Development Report for Latin america and the Caribbean. Multidimensional progress : well-being beyond income*. New York.

United Nations Environment Programme (UNEP). (2016). *GLOBAL ENVIRONMENT OUTLOOK. GEO-6. REGIONAL ASSESSMENT FOR LATIN AMERICA AND THE CARIBBEAN*. Nairobi, Kenya. Retrieved from http://prensa.usal.edu.ar/archivos/prensa/docs/GEO6.pdf

United Nations Environmental Program (UNEP). (2008). *UNEP 2007 Annual Report*.

United Nations Environmental Program (UNEP). (2010). *Latin America and the Caribbean: Environmental outlook*. *Relations between Environmental Changes and Human WellBeing in Latin America and the Caribbean*. Retrieved from http://www.unep.org/pdf/GEOLAC_3_ENGLISH.pdf

United Nations Oceans and the law of the sea. *First Global Integrated Marine Assessment (First World Ocean Assessment), 1833 United Nations Treaty Series § (2016).*

United Nations Population Division. (2014). *World Urbanization Prospects: The 2014 Revision, Highlights (ST/ESA/SER.A/352).*

United Nations-Department of Economic and Social Affairs- Population Division. (2015). *World Population Prospects*. Retrieved August 25, 2017, from https://esa.un.org/unpd/wpp/Download/Standard/Population/

UN-WATER. (2013). *Water Security & the Global Water Agenda*. Retrieved from

https://collections.unu.edu/view/UNU:2651
UN-WATER. (2016). *The United Nations World Water Development Report 2016: Water and Jobs*. Paris.
Uprety, Y., Asselin, H., Dhakal, A., & Julien, N. (2012). Traditional use of medicinal plants in the boreal forest of Canada: review and perspectives. *Journal of Ethnobiology and Ethnomedicine*, *8*(1), (pp. 7). https://doi.org/10.1186/1746-4269-8-7
Urbani, B. (2005). *The targeted monkey: a re-evaluation of predation on New World primates. J Anth Sci*, *83*, (pp. 89–109). Retrieved from http://www.isita-org.com/jass/contents/2005 vol83/urbani.pdf
Urbani, B., & Cormier, L. A. (2015). The Ethnoprimatology of the Howler Monkeys (Alouatta spp.): From Past to Present. In M. M. Kowalewski, P. A. Garber, L. Cortés-Ortiz, B. Urbani, & D. Youlatos (Eds.), *Howler Monkeys: Behavior, Ecology, and Conservation* (pp. 259–280). New York, NY: Springer New York. https://doi.org/10.1007/978-1-4939-1960-4_10
USDA FAS. (2015). Livestock and Poultry: World Markets and Trade.
USDA. (2014). *Uruguay, Livestock and Products, Annual 2014*. *Gain report*. Washington.
USDA. (2016). *World Agricultural Supply and Demand Estimates*. *United States Department of Agriculture*. https://doi.org/WASDE-525
USDA. (2017). *Animal Products.* Retrieved December 1, 2017, from https://www.ers.usda.gov/topics/animal-products/cattle-beef/statistics-information.aspx
USEIA. (2014). Increase in wood as main source of household heating most notable in the Northeast. Retrieved from https://www.eia.gov/todayinenergy/detail.php?id=15431
Usher, P. J. (2002). *Inuvialuit use of the Beaufort Sea and its resources, 1960-2000*. *Arctic*, *55*(SUPPL. 1), (pp. 18–28). https://doi.org/10.2307/40512417
Usher, R. (2002). *Putting Space Back on the Map: globalisation, place and identity. Educational Philosophy and Theory*, *34*(1), (pp. 41–55). https://doi.org/10.1111/j.1469-5812.2002.tb00285.x
Valdez, R., & Krausman, P. R. (1999). *Mountain Sheep of North America*. University of Arizona Press.
Van der Ploeg, S., & de Groot, R. S. (2010). *The TEEB Valuation Database – a searchable database of 1310 estimates of monetary values of ecosystem services*. Wageningen, The Netherlands.
Van Tussenbroek, B. I., Cort??s, J., Collin, R., Fonseca, A. C., Gayle, P. M. H., Guzm??n, H. M., J??come, G. E., Juman, R., Koltes, K. H., Oxenford, H. A., Rodr??guez-Ramirez, A., Samper-Villarreal, J., Smith, S. R., Tschirky, J. J., & Weil, E. (2014). Caribbean-wide, long-term study of seagrass beds reveals local variations, shifts in community structure and occasional collapse. *PLoS ONE*, *9*(3). https://doi.org/10.1371/journal.pone.0090600
van Zanten, B. T., van Beukering, P. J. H., & Wagtendonk, A. J. (2014). Coastal protection by coral reefs: A framework for spatial assessment and economic valuation. *Ocean & Coastal Management*, *96*, (pp. 94–103). https://doi.org/https://doi.org/10.1016/j.ocecoaman.2014.05.001
Vanacker, V., Vanderschaeghe, M., Govers, G., Willems, E., Poesen, J., Deckers, J., & De Bievre, B. (2003). Linking hydrological, infinite slope stability and land-use change models through GIS for assessing the impact of deforestation on slope stability in high Andean watersheds. *Geomorphology*, *52*(3–4), (pp. 299–315). https://doi.org/10.1016/S0169-555X(02)00263-5
Vandevijvere, S., Chow, C. C., Hall, K. D., Umali, E., & Swinburn, B. A. (2015). Increased food energy supply as a major driver of the obesity epidemic: a global analysis. *Bulletin of the World Health Organization*, *93*(7), (pp. 446–456). https://doi.org/10.2471/BLT.14.150565
Vasisht, K., Sharma, N., & Karan, M. (2016). Current Perspective in the International Trade of Medicinal Plants Material: An Update. *Current Pharmaceutical Design*, *22*(27), (pp. 4288–4336). https://doi.org/10.2174/1381612822666160607070736
Vazquez, P., Zulaica, L., & Benevidez, B. (2017). Agriculturización e Impactos Ambientales en el Partido de Necochea, Provincia de Buenos Aires, Argentina. *Raega - O Espaço Geográfico Em Análise*, *39*, (pp. 202–218). https://doi.org/10.5380/raega
Veldman, J. W., Overbeck, G. E., Negreiros, D., Mahy, G., Le Stradic, S., Fernandes, G. W., Durigan, G., Buisson, E., Putz, F. E., & Bond, W. J. (2015). Where Tree Planting and Forest Expansion are Bad for Biodiversity and Ecosystem Services. *BioScience*, *65*(10), 1011–1018. https://doi.org/10.1093/biosci/biv118
Verhoeven, J., Arheimer, B., Yin, C., & Hefting, M. (2006). Regional and global concerns over wetlands and water quality. *Trends in Ecology & Evolution*, *21*(2), (pp. 96–103).

https://doi.org/10.1016/j.tree.2005.11.015

Verma, S., Kanwar, V. S., John, S., S., V., Kanwar, V. S., & John, S. (2015). *Water Supply Engineering*. New Delhi: Vikas Publishing House PVT LDT.

Viers, J. H., Williams, J. N., Nicholas, K. A., Barbosa, O., Kotzé, I., Spence, L., Webb, L. B., Merenlender, A., & Reynolds, M. (2013). Vinecology: pairing wine with nature. *Conservation Letters*, *6*(5), n/a-n/a. https://doi.org/10.1111/conl.12011

Viesmann, W. J., & Hammer, M. J. (2008). *Water Supply and Pollution Control* (Eight Edit). Saddle River, NJ: Pearson.

Villagra, P., Rojas, C., Ohno, R., Xue, M., & Gómez, K. (2014). A GIS-base exploration of the relationships between open space systems and urban form for the adaptive capacity of cities after an earthquake: The cases of two Chilean cities. *Applied Geography*, *48*, (pp. 64–78). https://doi.org/10.1016/j.apgeog.2014.01.010

Villanueva-Gutierrez, R., Roubik, D. W., Colli-Ucan, W., Guemez-Ricalde, F. J., Buchmann, S. L., Villanueva-Gutiérrez, R., Roubik, D. W., Colli-Ucán, W., Güemez-Ricalde, F. J., & Buchmann, S. L. (2013). A critical view of colony losses in managed Mayan honey-making bees (Apidae: Meliponini) in the heart of Zona Maya. *Journal of the Kansas Entomological Society*, *86*(4), 352–362. https://doi.org/10.2317/jkes130131.1

Vit, P., Roubik, D. W., & Pedro, S. R. M. (2013). Pot-Honey: A legacy of stingless bees. *Pot-Honey: A Legacy of Stingless Bees*, (pp. 1–654). https://doi.org/10.1007/978-1-4614-4960-7

Vittor, A. Y., Gilman, R. H., Tielsch, J., Glass, G., Shields, T., Lozano, W. S., Pinedo-Cancino, V., & Patz, J. A. (2006). The effect of deforestation on the human-biting rate of Anopheles darlingi, the primary vector of falciparum malaria in the Peruvian Amazon. *American Journal of Tropical Medicine and Hygiene*, *74*(1), 3–11. https://doi.org/74/1/3 [pii]

Vörösmarty, C. J., Hoekstra, A. Y., & Bunn, S. E. (2015). What scale for water governance? *Science*, *349*(6247), (pp. 478–478). https://doi.org/10.1126/science.349.6247.478-a

Vörösmarty, C. J., McIntyre, P. B., Gessner, M. O., Dudgeon, D., Prusevich, a, Green, P., Glidden, S., Bunn, S. E., Sullivan, C. a, Liermann, C. R., & Davies, P. M. (2010). Global threats to human water security and river biodiversity. *Nature*, *467*(7315), 555–561. https://doi.org/10.1038/nature09549

Voss, R., Quaas, M. F., Schmidt, J., & Kapaun, U. (2015). Ocean acidification may aggravate social-ecological trade-offs in coastal fisheries. *PLoS ONE*, *10*(3), (pp. 1–8). https://doi.org/10.1371/journal.pone.0120376

Vuille, M., Francou, B., Wagnon, P., Juen, I., Kaser, G., Mark, B. G., & Bradley, R. S. (2008). Climate change and tropical Andean glaciers: Past, present and future. *Earth-Science Reviews*, *89*(3–4), (pp. 79–96). https://doi.org/10.1016/j.earscirev.2008.04.002

Wall, D. H., & Six, J. (2015). Give soils their due. *Science*, *347*(6223), 695 LP-695. Retrieved from http://science.sciencemag.org/content/347/6223/695.abstract

Wall, D. H., Nielsen, U. N., & Six, J. (2015). Soil biodiversity and human health. *Nature*, *528*(7580), (pp. 69–76). Retrieved from http://dx.doi.org/10.1038/nature15744

Wallmo, O. C. (1981). *Mule and Black-tailed Deer of North America: A wildlife management institute book*. University of Nebraska Press.

Walsh, J. F., Molyneux, D. H., & Birley, M. H. (1993). Deforestation: effects on vector-borne disease. *Parasitology*, *106*(S1), S55–S75. https://doi.org/10.1017/S0031182000086121

Walter, K. S., & Gillett, H. J. (1998). *1997 IUCN red list of threatened plants*. Gland, Switzerland: IUCN - The World Conservation Union.

Wassenaar, T., Gerber, P., Verburg, P. H., Rosales, M., Ibrahim, M., & Steinfeld, H. (2007). Projecting land use changes in the Neotropics: The geography of pasture expansion into forest. *Global Environmental Change*, *17*(1), (pp. 86–104). https://doi.org/10.1016/j.gloenvcha.2006.03.007

WCPA. (2012). Natural Solutions: Protected areas maintaining essential water supplies. Fact sheet. Gland, Switzerland: IUCN/World Commission on Protected Areas.

Webb, J., Coomes, O. T., Mainville, N., & Mergler, D. (2015). Mercury Contamination in an Indicator Fish Species from Andean Amazonian Rivers Affected by Petroleum Extraction. *Bulletin of Environmental Contamination and Toxicology*, *95*(3), (pp. 279–285). https://doi.org/10.1007/s00128-015-1588-3

Webber, L., Kilpi, F., Marsh, T., Rtveladze, K., Brown, M., & Mcpherson, K. (2012). High Rates of Obesity and Non-Communicable Diseases Predicted across Latin America. *PLoS ONE*, *7*(8), (pp. 1–6). https://doi.org/10.1371/journal.pone.0039589

Weeratunge, N., Béné, C., Siriwardane, R., Charles, A., Johnson, D., Allison, E. H., Nayak, P. K., & Badjeck, M.-C. (2014). Small-scale fisheries through the wellbeing lens. *Fish and Fisheries*, *15*(2), 255–279. https://doi.org/10.1111/faf.12016

Wentz, D. A., Brigham, M. E., Chasar, L. C., Lutz, M. A., & Krabbenhoft, D. P. (2014). *Mercury in the Nation ' s Streams — Levels , Trends , and Implications*. Reston, Virginia. https://doi.org/10.3133/cir1395

Werden, L., Barker, I. K., Bowman, J., Gonzales, E. K., Leighton, P. A., Lindsay, L. R., & Jardine, C. M. (2014). Geography, Deer, and Host Biodiversity Shape the Pattern of Lyme Disease Emergence in the Thousand Islands Archipelago of Ontario, Canada. *PLoS ONE*, *9*(1), e85640. https://doi.org/10.1371/journal.pone.0085640

Weyland, F. and Zaccagnini, M.E. (2008) Efecto de las terrazas sobre la diversidad de artrópodos caminadores en cultivos de soja. Ecología Austral 18: 357-366.

White, C. L., & Thompson, J. (1955). The Llanos: A Resource Neglected. *Journal of Range Management*, *8*, 11–17.

White, R., Murray, S., & Rohweder, M. (2000). *Pilot Analysis of Global Ecosystems: Grassland Ecosystems. World Resources Institute*. https://doi.org/10.1021/es0032881

White, R.G. Thomson, B.R. Skogland, T. Person, S.J. Russell, D.E. Holleman, D.F. Luick, J. R. (1975). Ecology of caribou at Prudhoe Bay, Alaska. In J. Brown (Ed.), *Ecological Investigations of the Tundra Biome in theprudhoe Bay Region, Alaska.* (pp. 150–210). Fairbanks, Alaska: University of Alaska.

Whiteman, A., Wickramasinghe, A., & Piña, L. (2015). Global trends in forest ownership, public income and expenditure on forestry and forestry employment. *Forest Ecology and Management*, *352*, (pp. 99–108). https://doi.org/10.1016/j.foreco.2015.04.011

Wilcox, B. A., & Ellis, B. (2006). Forests and emerging infectious diseases of humans. *Unasylva*, *57*(224), (pp. 11–18).

Willer, H. and Lernoud, J. (Eds). (2017). *The World of Organic Agriculture. Statistics and Emerging Trends*. Bonn, Germany.

Willis, K. J. (ed. . (2017). *State of the World's Plants 2017*. Royal Botanic Gardens Kew. https://doi.org/978-1-84246-628-5

Wilson, D. E., & Ruff, S. (1999). *The Smithsonian Book of North American Mammals*. (D. E. Wilson & S. Ruff, Eds.). Vancouver, BC: University of British Columbia Press.

Wilson, E. O. (1988). *Biodiversity*. *Biodiversity*. *National Academies Press*. https://doi.org/10.1111/j.1523-1739.1990.tb00309.x

Wittman, H., Chappell, M. J., Abson, D. J., Kerr, R. B., Blesh, J., Hanspach, J., Perfecto, I., & Fischer, J. (2017). A social–ecological perspective on harmonizing food security and biodiversity conservation. *Regional Environmental Change*, *17*(5), 1291–1301. https://doi.org/10.1007/s10113-016-1045-9

Wood, C. L., Lafferty, K. D., DeLeo, G., Young, H. S., Hudson, P. J., & Kuris, A. M. (2014). Does biodiversity protect humans against infectious disease? *Ecology*, *95*(4), (pp. 817–832). https://doi.org/10.1890/13-1041.1

Woodward, R. T., & Wui, Y.-S. (2001). The economic value of wetland services: a meta-analysis. *Ecological Economics*, *37*(2), (pp. 257–270). https://doi.org/https://doi.org/10.1016/S0921-8009(00)00276-7

World Bank (2017a). World Development Indicators. Indicator: Energy Use Per Capita. https://data.worldbank.org/indicator (Energy & Mining – Energy use)

World Bank 2017. World Development Indicators, https://data.worldbank.org/data-catalog/ world-development-indicators. Renewable internal freshwater resources per capita (cubic meters). Last updated: January 3, 2017.

World Bank, IUCN, ESA, PWA. (2010). Capturing and conserving natural coastal carbon: Building mitigation, advancing adaptation. World Bank/ IUCN/ESA PWA.

World Bank. (2017b). Terrestrial protected areas (% of total land area). 1990-2016. Retrieved December 15, 2018, from http://data.worldbank.org/indicator/ER.LND.PTLD.ZS

World Bank. (2017c). Terrestrial and marine protected areas (% of total territorial area). Retrieved

December 15, 2017, from https://data.worldbank.org/indicator/ER.PTD.TOTL.ZS

World Cancer Research Fund International. (2014). *The link between food , nutrition , diet and non-communicable diseases. London: World Cancer Research Fund International and the NCD Alliance.*

World Health Organization & Secretariat of the Convention on Biological Diversity. (2015). *Connecting Global Priorities : Biodiversity and Human Health A state of Knowledge Review*. Geneva Switzerland. Retrieved from www.who.int

World Health Organization (WHO). (2002). *WHO Traditional Medicine Strategy 2002-2005*. *World Health Organisation Geneva*. Geneva Switzerland: World Health Organization (WHO). https://doi.org/WHO/EDM/TRM/2002.1

World Health Organization (WHO). (2008). *World Water Day Report*. Geneva Switzerland. Retrieved from http://www.who.int/water_sanitation_health/takingcharge.html

World Health Organization (WHO). (2013). *WHO Traditional Medicine Strategy 2014-2023*. Geneva Switzerland. https://doi.org/2013

World Health Organization (WHO). (2016a). *Achieving the 2030 Target, 2030*. Retrieved from http://www.who.int/gho/publications/world_health_statistics/2016/whs2016_AnnexA_Sexual ReprodHealth.pdf. Accessed on 22nd September, 2016

World Health Organization (WHO). (2016c). *MCEE-WHO methods and data sources for child causes of death 2000-2015*. *Global Health Estimates Technical Paper* (Vol. 1). Retrieved from http://www.who.int/healthinfo/global_burden_disease/en/

World Health Organization (WHO). (2017). *Global Health Observatory Data Repository*. Retrieved October 24, 2017, from //apps.who.int/ghoen/data/view.main.CTRY2450A?lang=

World Health Organization. (2006). *WHO Air quality guidelines for particulate matter, ozone, nitrogen dioxide and sulfur dioxide: global update 2005: summary of risk assessment*. *Geneva: World Health Organization*, (pp. 1–22). https://doi.org/10.1016/0004-6981(88)90109-6

World Meteorological Organization. (2012). The State of Greenhouse Gases in the Atmosphere Based on Global Observations through 2011. *GREENHOUSE GAS BULLETIN*, *8*.

Worm, B., Barbier, E. B., Beaumont, N., Duffy, J. E., Folke, C., Halpern, B. S., Jackson, J. B., Lotze, H. K., Micheli, F., & Palumbi, S. R. (2006). Impacts of biodiversity loss on ocean ecosystem services. *Science*, *314*(5800), 787–790.

Wortley, L., Hero, J.-M., & Howes, M. (2013). Evaluating Ecological Restoration Success: A Review of the Literature. *Restoration Ecology*, *21*(5), (pp. 537–543). https://doi.org/10.1111/rec.12028

Wouters, P., & Tran, T. (2011). Out of the Mainstream: Water Rights, Politics and Identity. *Mountain Research and Development*, *31*(3), (pp. 270–271). https://doi.org/10.1659/mrd.mm081

Wunder, S. (2005). *Payments for environmental services : Some nuts and bolts* (No. 42). *CIFOR Occasional Paper* (Vol. 42). Jakarta, Indonesia. https://doi.org/10.1111/j.1523-1739.2006.00559.x

WWAP (United Nations World Water Assessment Programme). (2009). *The United Nations World Water Development Report 3: Water in a Changing World*. Paris.

WWAP (United Nations World Water Assessment Programme). (2015). *The United Nations World Water Development Report 2015: Water for a Sustainable World*. Paris: UNESCO.

WWF. (2016). *Living planet report: risk and resilience in a new era*. *WWF International*. Gland, Switzerland. Retrieved from http://awsassets.panda.org/downloads/lpr_living_planet_report_2016.pdf%0Ahttp://www.footprintnetwork.org/documents/2016_Living_Planet_Report_Lo.pdf

Zaccagnini. M.E., M. G. Wilson & J. D. Oszust. Eds. (2014) Manual de Buenas Prácticas para la Conservación del Suelo, la Biodiversidad y sus Servicios Ecosistémicos. 1ª ed. Buenos Aires. Programa Naciones Unidas para el Desarrollo -PNUD-SAyDS-INTA. 95 pp. https://inta.gob.ar/sites/default/files/inta-manual-de-buenas-practicas-para-la-conservacion-del-suelo-la-biodiversidad.pdf

Zenteno-Brun, H. (2009). Acercamiento a la visión cósmica del mundo Andino. *Punto Cero*, *14*(18), (pp. 83–89). Retrieved from http://www.scielo.org.bo/scielo.php?script=sci_arttext&pid=S1815-02762009000100010&lng=es&nrm=iso

Zimmerman, J. B., Mihelcic, J. R., & Smith, and J. (2008). Global Stressors on Water Quality and Quantity. *Environmental Science & Technology*, *42*(12), (pp. 4247–4254). https://doi.org/10.1021/es0871457

Chapter 3: Status and trends of biodiversity and ecosystem functions underpinning nature's benefit to people

Coordinating Lead Authors:
Jeannine Cavender-Bares (USA); Mary T. K. Arroyo (Chile)

Lead Authors:
Robin Abell (USA); David Ackerly (USA); Daniel Ackerman (USA); Matias Arim (Uruguay); Jayne Belnap (USA); Francisco Castañeda Moya (Guatemala); Laura Dee (USA); Natalia Estrada-Carmona (Colombia/France); Judith Gobin (Trinidad and Tobago); Forest Isbell (USA); Gunther Köhler (Germany); Marten Koops (Canada); Nathan Kraft (USA); Nicholas Macfarlane (Canada); Cristina Martínez-Garza (Mexico); Jean-Paul Metzger (Brazil); Arturo Mora (Ecuador); Mike Oatham (Trinidad and Tobago); Adriano Paglia (Brazil); Julieta Pedrana (Argentina); Pablo Luis Peri (Argentina); Gervasio Piñeiro (Uruguay); Robert Randall (Canada); Wren Walker Robbins (USA); Judith Weis (USA); Silvia Renate Ziller (Brazil).

Fellow:
Rodolfo Jaffe (Brazil/Venezuela)

Contributing Authors:
Marcelo Aizen (Argentina); Diva Amon (Trinidad and Tobago); Manuel Arroyo-Kalin (UK); Abigail Barker (UK); Keith Barker (USA); Darcy Bradley (USA); Kate Brauman (USA); Eduardo S. Brondizio (Brasil); Jarrett Byrnes (USA); Chris Caldwell (USA); Alejandro Casas (Mexico); Kenneth G. Cassman (USA); Joel Cracraft (USA); Frank Davis (USA); Juan Dupuy (Mexico); Brian Enquist (USA); Beth Fallon (USA); Curtis Flather (USA); Lee Frelich (USA); Susan Galatowitsch (USA); Kelly Garbach (USA); Jeff Grignon (Menominee Nation, USA); Hanno Seebens (Germany); Sharon Jansa (USA); Jon Keeley (USA); Christina Kennedy (USA); Kenneth Kozak (USA); Ulrike Krauss (Saint Lucia); Ian McFadden (USA); Mike Melnychuk (USA); Christian Messier (Canada); Ana Isabel Moreno-Calles (Mexico); Mark Nelson (USA); Danilo Mesquita Neves (Brazil); Cathleen Nguyen (USA); Mary O'Connor (Canada); Josep Padullés (Spain); Shyama Pagad (New Zealand); Alain Paquette (Canada); Lindsey Peavey (USA); Jesus Pinto-Ledezma (Bolivia); Diana Ramírez-Mejía (Mexico); María Francisca Flores Saavedra (Chile); Jane Smith (USA); Valeria Souza (Mexico); Katharine Suding (USA); Hans ter Steege (Holland/Brazil); Wolke Tobón (Mexico); Ignacio Torres-García (Mexico); Mark van Kleunen (Germany).

Review Editors: Patricia Balvanera (Mexico), Alexis Cerezo (Guatemala).

To be cited as:
Cavender-Bares, J., Arroyo, M.T.K., Abell, R., Ackerly, D., Ackerman, D., Arim, M., Belnap, J., Castañeda Moya, F., Dee, L., Estrada-Carmona, N., Gobin, J., Isbell, F., Köhler, G., Koops, M., Kraft, N., Mcfarlane, N., Martínez-Garza, C., Metzger, J. P., Mora, A., Oatham, M., Paglia, A., Pedrana, J., Peri, P. L., Piñeiro, G., Randall, R., Robbins, W. W., Weis, J., and Ziller, S. R. Chapter 3 Status and trends of biodiversity and ecosystem functions underpinning nature's benefit to people. In IPBES (2018): The IPBES regional assessment report on biodiversity and ecosystem services for the Americas. Rice, J., Seixas, C. S., Zaccagnini, M. E., Bedoya-Gaitán, M., and Valderrama, N. (eds.). Secretariat of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services, Bonn, Germany, pp. xx-xx

Table of contents Chapter 3

3 Executive Summary

1. **The Americas house a large fraction of the Earth's terrestrial and freshwater biodiversity distributed across 140 degrees of latitude (*well established*).** Around 29 per cent of the world´s seed plants, 35 per cent of mammals, 35 per cent of reptiles, 41 per cent of birds and 51 per cent of amphibians are found in the Americas (*established but incomplete*) {3.2.2.2}, as well as the world´s most diversified freshwater fish fauna of over 5,000 species (*well established*) {3.2.3.1}. The South American subregion is by far the richest subregion for plants and vertebrates (*well established*) {3.2.2.2}. However, the smaller Caribbean and Mesoamerican subregions are very rich for their areas, and North America contains both biodiversity hotspots and unique lineages {3.2.2.2}. The moist tropical lowland forests and tropical high Andean ecosystems contain high biodiversity on a global scale (*well established*) {3.4.1.1, 3.4.1.5}. Numbers of species and total evolutionary distance are generally higher in the tropics, while evolutionary distinctiveness tends to be higher in temperate latitudes {3.2.2.2}. Phylogenetic endemism is important for different taxa in different regions, and geographic patterns of plant functional diversity depend on the trait considered {3.2.2.2}. Biodiversity in all subregions has conservation significance {3.2.2.2} and all biomes provide nature´s contributions to people; the five most important terrestrial biome contributors are: Tropical and subtropical moist forests; Temperate and boreal forests and woodlands; Tropical and subtropical dry forests; Mediterranean forests, woodlands and scrub; Tundra and high elevation habitats (*established but incomplete*) {3.4.1.10}. For aquatic systems, freshwater habitats stand out (*established but incomplete*) {3.4.1.10}.

2. **The biodiverse American tropics became a major center of origin for domesticated plants (*well established*) and of traditional agriculture.** Many plants domesticated in Mesoamerica, the Andean region, and the Amazon Basin have become important crops globally (*well established*) {3.3.3, 3.4.1.1, 3.4.1.5}. Traditional agricultural systems harbor high levels of biodiversity and represent a high-quality matrix that allows forest species movements among patches (*established but incomplete*) {3.3.3}. Traditional farming systems have a structural complexity and multifunctionality that benefit people and ecosystems; they allow farmers to maximize harvest security and reap the benefits of multiple use of landscapes with lower environmental and biodiversity impacts (*established but incomplete*) {3.3.3}.

3. **Many terrestrial biomes, or large parts thereof, in the Americas have lost around 50 per cent or more of habitat, leading to losses in biodiversity and ecosystem functions (*well established*). A few biomes, however, are now showing recuperation or are fairly stable (*established but incomplete*)**. Close to 50 per cent of the Great Plains grasslands, including over 95 per cent of tallgrass prairie; some 88 per cent of the south atlantic forest; nearly 70 per cent of the South American Río de la Plata grasslands; 82 per cent of mesic broadleaf forest in Mexico; 72 per cent of tropical and dry forest in Mesoamerica; 66 per cent of tropical dry forest in the Caribbean; 50 per cent of the broader South American Mediterranean-climate biome; and 50 per cent of Cerrado has been transformed, mostly ongoing, leading to declines in native species richness and population sizes and nature's contributions to people (*well established*) {3.4.1.1, 3.4.1.2, 3.4.1.4, 3.4.1.6, 3.4.1.7, 3.4.1.10}. Notwithstanding a perceptible trend for conversion of páramo and puna in some parts of the northern Andes, the tundra and high elevation habitat biome is the least transformed {3.4.1.10}. Agriculture and deforestation have led to depletion of soil organic carbon, lowering of carbon stocks and affected the water cycle (*established but incomplete*) {3.2.1.1, 3.2.1.2}. Presently Caribbean forests are expanding (*well established*) {3.2.2.1, 3.4.1.1} and North American forests are stable to slightly increasing (*established but incomplete*) {3.2.2.1}.

4. **Experimental evidence and empirical observations support linkages between biodiversity and ecosystem productivity, stability and resistance to stress (*well established*).** A large number of studies across taxonomic groups and biomes (temperate and tropical forests, grasslands and marine systems) show greater productivity, stability, and stress resistance of ecosystems with higher biodiversity {3.1.2, 3.1.3}, indicating that biodiversity is relevant to sustainability. The majority of studies within the Americas were conducted in North America, but studies in Mesoamerica and South America are consistent with results for North America and global findings.

5. **The transformation of wetlands in the Americas has led to loss of biodiversity (*established but incomplete*) and ecosystem functions (*well established*).** From 1976 to 2008, the Brazilian pantanal experienced a huge loss of floodplains (*well established*) affecting biodiversity (*established but incomplete*) {3.4.1.9}. One-third of the freshwater marshes in the lower Paraná delta were converted between 1999 and 2013 (*well established*) {3.4.1.9}. The vast biologically rich South American Pantanal has been increasingly degraded due to cattle ranching and cropping (*well established*) {3.4.1.9}. Mechanized peat mining in southern temperate peatlands has promoted invasive plant species, increased beaver presence and produced hydrological changes (*well established*) {3.4.1.9}. In recent years, the United States of America lost an average of 5,600 hectares per year of wetland habitat, lowering capacity for water filtration {3.4.1.9}. In the past four decades, invasive species have become an increasing threat to biodiversity in the Florida Everglades and other wetlands (*established but incomplete*) {3.4.1.9}. Some wetlands in Mesoamerica have been contaminated with heavy metals and pesticides (*established but incomplete*) {3.4.1.9}.

6. **Oceans of the Americas contain high biodiversity, can have high numbers of threatened species, and include large numbers of species that are important for human well-being (*established but incomplete*)**. Respectively, over 12,000 marine organisms have been found in the Caribbean, 10,000 in the Humboldt Current system, and 9,000 on the Brazilian shelves {3.2.4.1}, numbers that are considered to be conservative. Oceans of the Americas contain three of the seven global threat hotspots for neritic and epipelagic oceanic sharks in coastal waters (*established but incomplete*) {3.4.2}. The highest number of threatened or endangered marine mammal stocks around the globe are found in the Pacific, but some populations have recently begun to recover (*well established*) {3.4.2}. Stock assessments for a number of chondrichthyes in the Americas report declines of 20 to 80 per cent from unfished conditions. In Canada, marine fish populations declined by an average of 52 per cent from 1970 to the mid-1990s and then remained stable (*established but incomplete*) {3.4.2}.

7. **Biodiversity in coastal habitats has experienced major losses in recent decades (*well established*).** Coral reefs in the Caribbean declined in cover by more than 50 per cent by the 1970s, with only 10 per cent remaining by 2003, followed by widespread coral bleaching in 2005 and subsequent mortality from infectious diseases (*established but incomplete*) {3.4.2.1}. Coastal salt marshes and mangroves are rapidly disappearing (*established but incomplete*) {3.4.2.1}. Considerable declines in seagrasses have occurred (*established but incomplete*) {3.4.2.1}.

8. **Urban expansion constitutes both a threat to biodiversity and an opportunity for biodiversity conservation {*established but incomplete*).** Urban areas are now home to 80 per cent of the population of the Americas {3.3.4}. Urban encroachment is associated with declining native species richness and shifts in species composition, yet increased total plant diversity with cultivation of non-native species (*established but incomplete*) {3.3.4}. Remnant vegetation in cities can support significant native biodiversity, such as bees and birds (*well established*).

Botanical gardens, major reservoirs of ex situ conservation, and important for recreation and environmental education, found mostly in urban areas, are unequally distributed among subregions and biomes (*well established*) {3.3.4}. Green areas that incorporate native biodiversity have the potential to accomplish the dual goals of conservation and human well-being{3.3.4}.

9. **Alien species continue to appear in terrestrial, freshwater, and marine habitats in the Americas, but rates of introduction, where known, differ among subregions (*established but incomplete*).** Terrestrial and marine habitats house outstanding numbers of alien plants and bird species {3.2.2.3, 3.2.4.2}. North America and the Caribbean are the mostly strongly invaded subregions (*established but incomplete*) {3.2.2.3}. Rates of appearance of alien species are currently somewhat lower in North America than in South America (*established but incomplete*) {3.2.2.3}. Marine habitats of the North American subregion are more heavily invaded than other subregions, with the Pacific Ocean more invaded than the Atlantic (*established by incomplete*) {3.2.4.2}. For freshwater, temperate piscivorous, and carnivorous fish cause negative impacts on the native fish fauna (*established but incomplete*) {3.2.3.2}. In the Americas, several endangered and threatened species have declined as a result of emerging infectious diseases {3.2.6}. Strongly invasive alien species can entail significant economic costs for infrastructure {3.2.3.2}, and significantly lower productivity (*well established*) {3.2.2.3}.

10. **Overall, species threat level is high in the Americas, but the underlying causes vary among subregions (*established but incomplete*).** Based on 14,000 species assessed that occur in the Americas, close to a quarter of species face extinction risk (*established but incomplete*) {3.5.1}. Aggregate threat risk over the past two decades was highest in South America and the Caribbean (*well established*) {3.5.1}. Since 1989, the number of threatened North American freshwater fishes has increased by 25 per cent, with 7.5 extinct taxa per decade post-1950 {3.2.3.1} (*well established*). In Central America, 42 per cent of close to 500 known amphibian species have been assessed as threatened (*well established*) {3.2.3.1}. The International Union for Conservation of Nature category "Invasive species, other problematic species, genes and diseases" is the main cause for extinction risk in the North American subregion, while the categories "Biological resources use" and "Agriculture and aquaculture" are the most important causes in the Mesoamerican, Caribbean and South American subregions (*established but incomplete*) {3.5.1}.

11. **While protection measures in the Americas have increased and diversified over the past 30 years, major differences in protection effort persist between terrestrial and marine ecosystems and among biomes (*well established*).** The increase in protection has been notable in South America where 25 per cent of this subregion is now protected. South America, Mesoamerica, and the Caribbean lag behind North America in terms of marine protection (*well established*) {3.5.2}. Twenty percent of all designated key biodiversity areas globally are found in the Americas (*well established*} {3.5.2}, yet, less than 20 per cent of these are completely covered (*well established*). Certain biomes are still poorly protected (*well established*) {3.5.2}. Temperate grasslands in general and South American Mediterranean forests, woodlands and scrub and drylands are among the least protected biomes {3.5.2}. Tropical and subtropical savanna and grasslands, tropical and subtropical dry forests, and tropical and subtropical coniferous forests are poorly protected {3.5.2}. Indigenous reserves and private initiatives and are increasingly important {3.5.2}.

12. **Many Aichi targets are unlikely to be met in some countries (*established but incomplete*).** Although the rate of loss of natural habitat has decreased in some biomes, degradation and

fragmentation continue {3.4.1.10}, making it unlikely to achieve Aichi target 5. Unsustainable fishing continues {3.4.2} (Aichi target 6). Likewise, many intensive agricultural, silvicultural and aquacultural systems do not follow biodiversity-friendly practices {3.3.5} (Aichi target 7). Alien and invasive alien species are widespread and continue to appear across the Americas {3.2.2.3, 3.2.3.2, 3.2.4.2, 3.4} (Aichi target 9). Coral bleaching continues in response to coastal pollution and global warming {3.4.2.1} (Aichi target 10). Total protected area coverage for the Americas is 14 per cent, with 18 per cent terrestrial and 9 per cent marine, but some biomes remain severely under-protected {3.5.2}. Better biome representation would allow meeting Aichi target 11. Although conservation efforts have improved, overall extinction risk for species has increased in some subregions {3.5.1} (Aichi target 12).

13. **Major biodiversity data and knowledge gaps persist across the Americas (*well established*).** Basic exploration is incomplete, especially in the richest biodiversity areas. Brazil contributed the largest number of new plant species to the global inventory from 2004 to 2016, and 42 per cent of recently described new mammals species worldwide between 1993 and 2008 came from the Americas (*well established*) {3.6}. In South America experts predict that around 50 per cent of marine biodiversity remains undiscovered (*established but incomplete*) {3.6}. Research on functional diversity and the relationship between biodiversity and ecosystem functions across taxonomic groups is growing but remains scarce in some subregions. Enormous data gaps persist at the biome level in all subregions. Despite its very high biodiversity, South American houses the fewest georeferenced species occurrence records per unit area, while the highest number is in North America, despite much lower richness {3.6}. Major challenges for the future are: scaling up from ecological studies to the biome level, coordinated conservation efforts in biomes that cross country boundaries, making all biodiversity data available online, and the production of standardized biodiversity data useful for policymakers {3.6}.

3.1 Background

3.1.1 Setting the stage

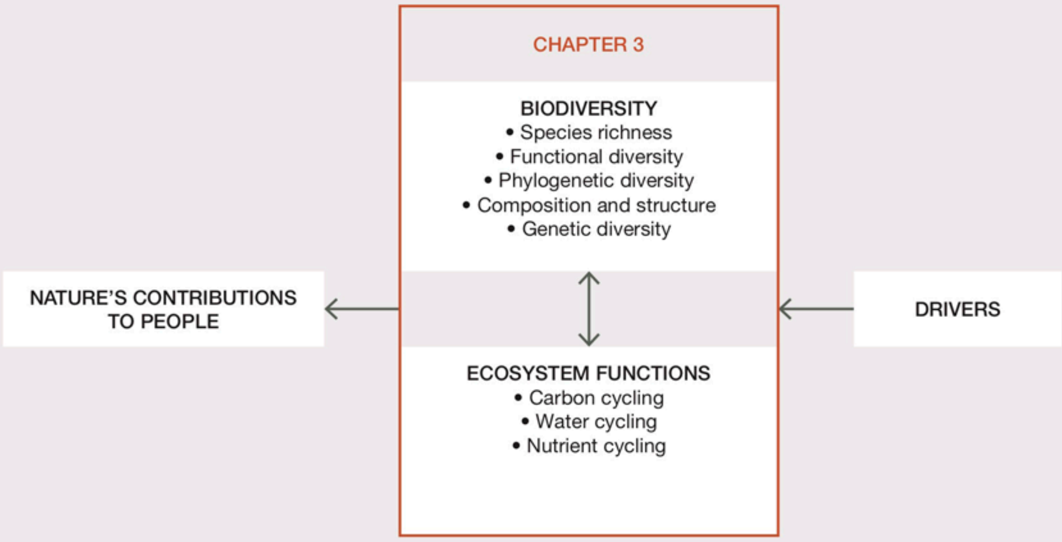
The Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) recognizes that humans benefit both consciously and unconsciously from ecosystem functions and biodiversity, through the ecosystem services they are coupled with, referred to as nature's contributions to people (NCP).

The biodiversity of the Americas comes from many different marine, freshwater, and terrestrial sources and offers humankind numerous products and services. To protect the enormous potential of this biodiversity to provide NCP, it is critical to understand the geographic distribution of biodiversity as well as how biodiversity, and the ecosystem functions that both depend on and support biodiversity, have been changing over time.

This chapter assesses: (1) our current understanding of the distribution, status and recent trends of ecosystem functions and biodiversity across the Americas; (2) how people interact with biodiversity, highlighting the importance of local and indigenous knowledge; (3) how biodiversity and ecosystem functions vary within and have changed across the units of analysis in each subregion; (4) current understanding of the extent to which biodiversity is imperiled and protected; and (5) major data and knowledge gaps in all of these realms. The chapter focuses on biodiversity and ecosystem function (Figure 3.1) in the context of how they contribute to NCP (Chapter 2) and are impacted by drivers of change (Chapter 4).

Figure 3 1 **Within the IPBES conceptual framework, Chapter 3 focuses on the status and trends of biodiversity, which encompasses a range of dimensions, and ecosystem functions, as well as the linkages between them.**

Biodiversity and ecosystem functions underpinning NCP, treated in Chapter 2, are influenced by drivers of change, treated in Chapter 4. See Chapter 1 for the complete IPBES framework. Source: own representation.



3.1.2 How is biodiversity linked to ecosystem functions and ecosystem services?

Biodiversity loss is known to substantially decrease ecosystem function and stability (Cardinale et al., 2011; O'Connor et al., 2017). Consequently, biodiversity loss and ecosystem degradation diminish the ability of humans to benefit from or establish spiritual relationships with other living beings.

The relationships between biodiversity and ecosystem function have been rigorously investigated in numerous experiments (e.g. Cardinale et al., 2011) and in theoretical (Loreau, 2010; Tilman et al., 1997) and observational studies in a wide range of ecosystems, including grasslands (Grace et al., 2016; Hautier et al., 2014), forests (Gamfeldt et al., 2013; Liang et al., 2016; Paquette & Messier, 2011), drylands (Maestre et al., 2012) and marine systems (Dee et al., 2016; Duffy et al., 2016), many conducted in the Americas. Recent studies have also revealed many potential benefits of increasing plant diversity in managed production systems, including enhancing the production of crops, forage, wood, and fish; stabilizing productivity; enhancing pollinators and pollination; suppressing weeds and other pests; and accumulating and retaining soil nutrients and carbon (Balvanera et al., 2006, 2014; Cardinale et al., 2012; Kremen & Miles, 2012; Letourneau et al., 2011; Quijas et al., 2010; Scherer-Lorenzen, 2014).

3.1.3 Conceptual and theoretical linkages between biodiversity and ecosystem functions and services

Biodiversity loss can alter ecosystem function. Here, we focus on relationships between plant diversity and productivity. Theory (Thébault & Loreau, 2003) and experiments (Lefcheck et al., 2015) have shown that these relationships are largely generalizable to other trophic levels. Furthermore, given that rates of primary productivity limit the energy available to animals at all higher trophic levels, effects of changes in biodiversity on productivity have many cascading effects on other pools and fluxes of matter and energy in ecosystems (McNaughton et al., 1989).

Plant species richness increases primary productivity when interspecific competition is reduced relative to intraspecific competition (Loreau, 2004; Vandermeer, 1981). Reduced competition among species for resources can occur in diverse communities because different plant species consume somewhat different resources (e.g. different forms of nitrogen) or consume the same resources at somewhat different times (e.g. phenological niche partitioning) or places (e.g. different rooting zones) (McKane et al., 2002; Tilman et al., 1997). Such resource partitioning likely contributes to both coexistence and positive effects of plant diversity on ecosystem productivity in many ecological communities (Turnbull et al., 2016). Similarly, increased plant species richness can lead to increased ecosystem productivity when there is reduced apparent competition in diverse communities because plant species can avoid natural enemies, such as specialized herbivores or pathogens, that become diluted in diverse communities (Petermann et al., 2008; Turnbull et al., 2016). Strong effects of complementarity between species or groups of species (Brooker et al., 2008), such as between grasses and legumes (Temperton et al., 2007), contribute to the positive effects of plant diversity on ecosystem productivity. Results from the five longest-running grassland biodiversity experiments suggest that these complementarity effects grow stronger over time, while the importance of individual species that are particularly productive become less important for ecosystem productivity (Fargione et al., 2007; Isbell et al., 2009; Marquard et al., 2009; Reich et al., 2012; van Ruijven & Berendse, 2009). Based on abundant empirical evidence, it is now well-established that local complementarity effects often explain positive effects of biodiversity on ecosystem productivity (Cardinale et al., 2011; Loreau & Hector, 2001), especially in long-term studies (Cardinale et al., 2007; Fargione et al., 2007; Isbell et al., 2009; Marquard et al., 2009; Reich et al., 2012; van Ruijven & Berendse, 2009), but the precise mechanisms are not always possible to discern and are the subject of ongoing research.

Biodiversity experiments address limitations of observational studies and have been designed and conducted to tease apart effects of changing numbers of species (richness) from effects of changing identities of species (composition) (c.f., O'Connor et al., 2017). Such experiments have revealed some surprisingly productive species and combinations of species, even when excluding legumes (van Ruijven & Berendse, 2005; Wilsey & Polley, 2004) or mixing species within functional groups (Bullock et al., 2007; Reich et al., 2004). Changes in grassland plant species richness can influence plant productivity as much as changes in species composition (Hector et al., 2011), intensive agricultural management (Weigelt et al., 2009) and many other factors long known to regulate plant productivity (Hooper et al., 2012; Tilman et al., 2012). Similar strengths of biodiversity effects on ecosystem function have been found in terrestrial and aquatic habitats (O'Connor et al., 2017). Meta-analysis reveals that herbivore diversity influences more ecosystem functions than plant diversity (Arias-González et al., 2016; Lefcheck et al., 2015). Additional examples of biodiversity links to ecosystem functions in different biomes and other units of analysis can be found throughout the chapter.

3.2 Continental distribution of ecosystem functions and biodiversity

3.2.1 Status and trends of ecosystem functions linked to biodiversity

3.2.1.1 Carbon cycling and energy fluxes

Status. The carbon cycle is strongly linked to land cover change (section 3.3.2) and energy flux since energy enters and moves through ecosystems in the form of carbon-based molecules. Therefore, the carbon cycle has major implications for ecosystem function and provisioning of ecosystem services. Land use change increases carbon emissions or sequestration depending on the nature of vegetation replacement. Agriculture and deforestation are the main land use changes that have altered carbon fluxes and stocks. Overall, agriculture has reduced carbon inputs to ecosystems through harvest and/or increased carbon output from cultivation; human appropriation of primary production (a measure of the amount of energy captured by humans from ecosystems) is particularly high in agricultural regions of the Americas (Haberl et al., 2007). Agricultural soils lose carbon when monocultures of annual crops are planted without rotations (Ernst & Siri-Prieto, 2009; Franzluebbers, 2005). However, recent trends in double cropping, no-till practices and used cover crops have the potential to at least partially restore soil organic carbon stocks (Franzluebbers, 2005; Poeplau et al., 2015; Rimski-Korsakov et al., 2016).

Forest ecosystems of Americas contain near 250 picograms of carbon (Köhl et al., 2015), with large amounts of biomass carbon stored in South American forests and high soil carbon stocks located in the permafrost boreal regions of Canada (Jackson et al., 2017). Deforestation (section 3.2.2.1 and Chapter 4) has significantly decreased plant biomass stocks (80 to 95%) throughout the Americas (Chapin et al., 2012) and also soil carbon stocks (Villarino et al., 2016) except in moist forests replaced by pastures that may increase soil organic carbon stocks (Eclesia et al., 2012; Guo & Gifford, 2002). Maintaining the integrity of forests in the Americas thus is essential for climate regulation. Croplands today in the Americas contain 20 to 40% less carbon than under native forest, savannas or grasslands (Alvarez, 2005; Guo & Gifford, 2002).

Recent trends. Forest regrowth in some parts of the North American subregion increased between 1990 and 2015 (Keenan et al., 2015), and primary production in plantation forests mostly in South America has sequestered significant atmospheric carbon (Wright et al., 2000). Recent decreases in deforestation rates in Amazonia have favored net atmospheric carbon sequestration (Davidson et al., 2012; Nepstad et al., 2014; Zarin et al., 2016). Afforestation of grasslands has increased carbon uptake (primary production) and biomass carbon stocks (Vassallo et al., 2013), and increased soil organic carbon on dry sites but decreased soil organic carbon contents on humid sites (Berthrong et al., 2012; Eclesia et al., 2012). While recent woody encroachment in the USA and Argentina has increased biomass stock, it may have negative impacts on deep carbon storage (Asner & Archer, 2010; Jackson et al., 2002). Satellite-detected trends in the Normalized Difference Vegetation Index (a proxy of primary production) support observed changes in carbon stocks (Hicke et al., 2002; Paruelo et al., 2004). Finally, oceans around the Americas must be storing significant amounts of carbon, given they represent a significant fraction of the 2 picograms/year global ocean total. The Americas total is not available.

The net impact of land use on climate change is still under debate (Anderson-Teixeira et al., 2012; Houspanossian et al., 2017; Jackson et al., 2008). Meanwhile, it is clear that soil organic carbon loss severely affects soil fertility and plant production and that such losses are associated with nutrient releases and erosion that promote the eutrophication of rivers, lakes and oceans, all affecting human well-being. Several studies show negative impacts of land use changes on water cycling and other ecosystem services (Jackson et al., 2005; Trabucco et al., 2008).

3.2.1.2 Water cycle and regulation

Status. The water cycle is strongly regulated by evapotranspiration, which reduces water available for runoff and groundwater recharge (Brauman et al., 2007). Evapotranspoiration depends on the physical structure of vegetation and characteristics of individual species, particularly rooting depth, which controls plant access to water in water-limited environments (Le Maitre et al., 2015). Woody vegetation generally has higher evapotranspiration than other vegetation, reducing streamflow (Bosch & Hewlett, 1982; Brown, et al., 2005; Sahin & Hall, 1996). Studies supporting this conclusion are largely from temperate regions (Andréassian, 2004), athough some research has also been carried out in the tropics (Cashman, 2014; Tomasella et al., 2009). The reduction in woody vegetation is also associated with higher soil infiltration (Farley et al., 2005; Ochoa-Tocachi et al., 2016). Changes in infiltration have been also attributed to the impact of conversion on soils, which are compacted by timber harvesting and cattle grazing (Tomasella, et al., 2009). The hydrologic impact of forest conversion to pasture depends on grazing intensity, with high-density grazing causing more surface flow (Ochoa-Tocachi et al., 2016). These kinds of links with biodiversity are important for water regulation.

Recent trends. Reduced evapotranspiration can lead to reduced rainfall. Measurements and models of climate impacts of deforestation demonstrate a threshold by which complete deforestation of the tropics would substantially reduce rainfall (Lawrence & Vandecar, 2015). More realistic measurements and models of deforestation in the Amazon and non-Amazonian South America, however, show land use change to reduce precipitation only on the order of a few percent (Lawrence & Vandecar, 2015). The impact of changing climate on streamflow is complex, and most large rivers worldwide have not changed measurably at this point. Ten of the 14 large rivers that show increasing discharge are in the Americas. These rivers mostly correspond to places where rainfall has measurably increased (Milliman et al., 2008).

3.2.1.3 Nutrient cycling

Status. Over the past century, land use change, new agricultural practices, and fossil fuel combustion have drastically disrupted nutrient cycles worldwide (Canfield et al., 2010). Latin America showed high biological nitrogen fixation in native ecosystems until the mid-1990s (26.6 teragrams of nitrogen) and maintained fertilization and legume crops at relatively low rates (5.0 and 3.2 teragrams of nitrogen, respectively). In contrast, North America is characterized by relatively low natural fixation (11.9 teragrams of nitrogen), and high fixation by legume crops (6.0 teragrams of nitrogen) and fertilization (18.3 teragrams of nitrogen) (Galloway et al., 2004). While increased nitrogen input into agricultural ecosystems in the Americas has increased food production, it has promoted a four-fold increase in river nitrogen exports and a four- to seven-fold increase in nitrogen emissions to the atmosphere (Galloway et al., 2004) resulting in reduced drinking water and air quality, freshwater eutrophication, biodiversity loss, rain acidification, stratospheric ozone depletion, climate change and coastal ecosystem destruction (dead zones). Severe pollution occurs with the discharge of the Mississippi River into the Gulf of Mexico, of several rivers on the eastern coast of North America and from some rivers in South America associated with agriculture (Diaz & Rosenberg, 2008).

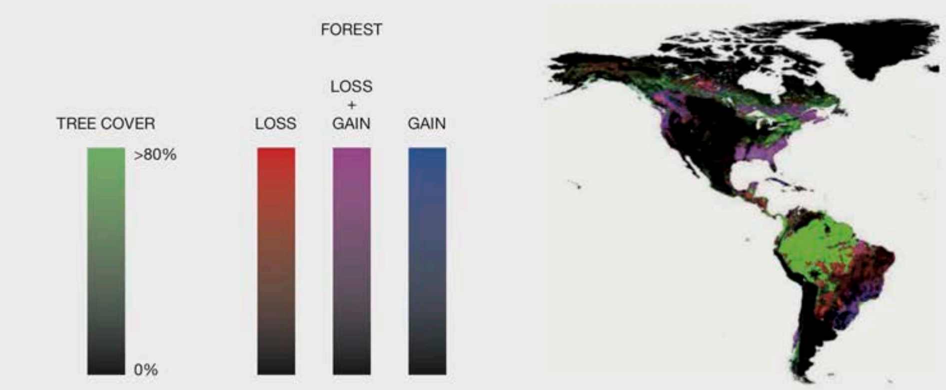
Recent trends. As a result of the green revolution, nitrogen inputs increased in the Americas, particularly in South America over the past two decades (Austin et al., 2006). Soybean crops expanded from 17 to more than 46 million ha between 1990 and 2010 (FAO, 2011). Some 48% of all croplands in southern South America (Brazil, Argentina, Uruguay, Paraguay, and Bolivia) are soybean (FAO, 2011). In addition, both North and South America have become key grain exporters; currently around 8 teragrams of nitrogen are being exported from the Americas, mainly to Europe and Asia, while around 6 teragrams of nitrogen come back as fertilizers, generating an imbalance in the region of 2 teragrams of nitrogen per year as a result of international trade (Galloway et al., 2008). However, the Americas show a better nutrient balance in agricultural systems than other regions of the world (Vitousek et al., 2009). Although technology is available for improved nutrient recycling in cities and farms, it is seldom used in the Americas (Grimm et al., 2015; Snapp at al., 1998). The use of legumes and catch crops (i.e. fast-growing crops that are grown between successive plantings of a main crop) to tighten or close the nitrogen cycle via synchronization of nutrient uptake and mineralization to avoid nutrient loses is a challenging issue for the Americas.

3.2.2 Status and trends of terrestrial biodiversity

3.2.2.1 Land cover status and trends

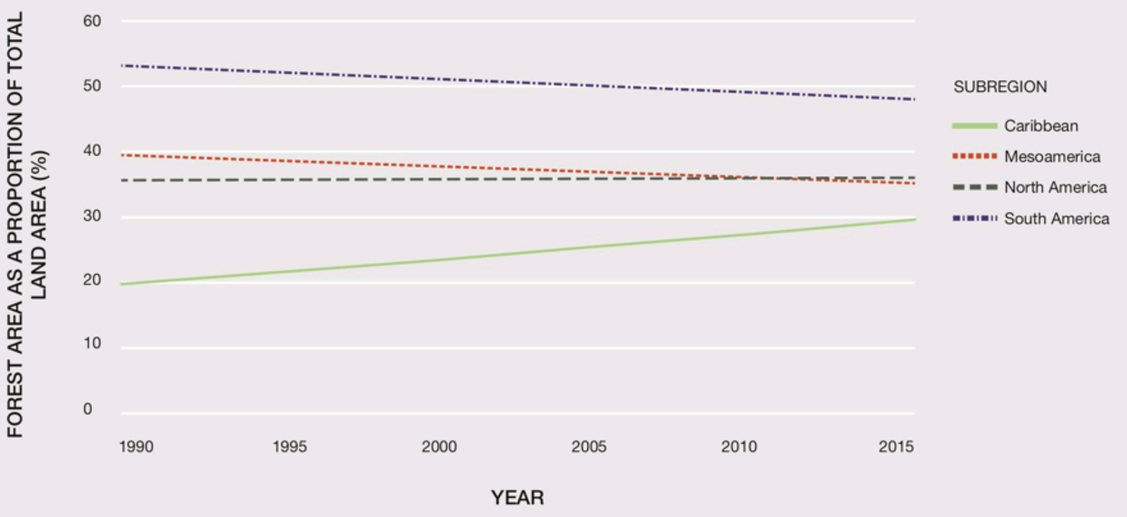
With better technology and availability of country surveys, we now have fairly reliable estimates of land cover in the Americas, especially for forests (Figures 3.2. and 3.3). More than two-thirds of the Americas is composed of closed to open vegetation, including forests, savannas, and grasslands, as well as mosaics of those vegetation types. About 16% of the region is occupied exclusively by croplands (e.g. corn, soybeans, wheat, sugarcane, and grazing land) and 1% by urban or bare land (Tuanmu & Jetz, 2014).

Figure 3 2 **Forest gain and loss across the Americas between 2000 to 2012. Source: Modified from Hansen *et al.* (2013).**



Forest cover in the Americas represents ca. 40% of the global forest cover, with ca. 842 millions of hectares in South America, 723 millions of hectares in North America and 20 millions of hectares in Central America (Keenan et al., 2015). Following the last update of Global Forest Watch (2017), which differentiates native from planted forests, the Americas have 1.668 millions of hectares of natural forest and ca. 67 millions of hectares of planted areas (e.g. timber, oil palm, rubber). Around 870 millions of hectares of the natural forest cover is considered primary forest (no clear indications of human activity or significant disturbance) and 797 millions of hectares is naturally regenerated native forests with clear indications of human activities (Global Forest Watch, 2017).

Figure 3 3 Total forest cover trends by subregions. Indicator data source: FAO (2015). The figure prepared by Task Group on Indicators and Knowledge and Data Technical Support Unit.



Forest cover has changed throughout the Americas in recent decades (Figure 3.3). It continues to decline in most subregions except in the Caribbean where forest regrowth predominates (see also 3.4.1.1). In North America the overall amount of forest has slightly increased (Figures 3.2 and 3.3). Further details on declines can be found for the specific biomes assessed in section 3.4.

Grasslands and shrublands are frequently confounded with agricultural areas or pasturelands at coarse scales and usually represented as a “mosaic of vegetation and cropland” (Arino et al., 2012). This mixed class covers about 12% of the Americas (and includes almost 80% of the croplands) (Arino et al., 2012) distributed predominantly in the USA (Central Great Plains), Canada (e.g. northern grasslands), Chile (Patagonian grasslands), Brazil (campos sulinos) and Argentina (pampas, Patagonian grasslands). Shrublands or savannas represent another 10% of the Americas’ land cover, with extensive coverage in the USA (e.g. Californian chaparral, arid shrublands, Great Plain shrublands) and Brazil (Cerrado). For details of changes in the different biomes of the Americas (section 3.4).

3.2.2.2 Status and patterns of diversity for taxonomic groups

Overall richness patterns. Despite several centuries of exploration, accessible and accurate data for biodiversity across the entire Americas is limited to a very small number of taxonomic groups. Data compiled at the subregional level for such groups confirms that the Americas region (comprising 28% of the world´s land area, including water bodies), holds significant proportions of the world’s biodiversity, as high as 51% for amphibians and 41% for birds (Table 3.1). Species richness is highest for all taxonomic groups in the South American subregion and far higher in South America than in North America (Table 3.1). Mesoamerica and the Caribbean are very rich in relation to their land area. For example, the Caribbean subregion (<1% of the Americas´ land area) is more diverse than North America (51% of the Americas´ area) for reptiles and is not that far behind for plants (Table 3.1). Mesoamerica (6% of the Americas´ land area) has more species in all taxonomic groups — in three out of five cases over twice as many — as the much larger North American subregion. The Americas account for some 33% of plants that have been recorded to be useful to humans globally (Table 3.1). The absolute numbers of useful plants in Table 3.1 are likely to be conservative, given that comprehensive surveys of useful plants have still to be undertaken in many parts of the Americas.

Table 3.1. Species richness for taxonomic groups where data could be compiled for IPBES subregions of the Americas. The percentages under the subregional headings give the amount of land (including water bodies) in relation to the total for the Americas. The percentages for the different taxonomic groups and useful plants in each subregion are calculated in relation to the totals for the Americas.

Taxon	Total for Americas	% of world total	North America (51%)	Mesoamerica (6%)	Caribbean (<1%)	South America (42%)
Plants[1,8] (seed plants only)	98,473 (108,320)[2]	29	13,214 (13%)	26,551 (27%)	11,473 (12%)	63,725 (65%)
Useful plants[3] (seed plants only)	10,188	33	4,252 (42%)	4,217 (41%)	2,915 (29%)	5,621 (55%)
Birds[4] – breeding species	4,374	41	649 (15%)	1,191 (27%)	320 (7%)	3205 (73%)
Mammals[5] native – terrestrial	1,963	35	458 (23%)	627 (32%)	185 (9%)	1,266 (64%)
Amphibians[6]	3,928	51	307 (8%)	812 (21%)	234 (6%)	2,809 (72%)
Reptiles[7]	3,652	35	431 (12%)	1231 (34%)	637 (17%)	1,990 (54%)

[1] Compiled by the Royal Botanic Gardens, Kew. Seed plants include angiosperms and gymnosperms and both native and non-native species. Data are from the World Checklist of Selected Plant Families (published and unpublished), which is 90% complete. The families Melastomataceae and Asteraceae and the genus *Solanum* are not included at this stage.

[2] Estimate if the two missing families and *Solanum* are included. Percentage of the world total in the Americas is based on the estimated total and a world total of seed plants of 370,492 (Lughadha et al., 2016). The subregional totals have not been adjusted and thus are conservative.

[3]The useful plant data come from the Royal Botanic Gardens, Kew Useful Plants Data Base. This database is formed from a combination of resources amounting to 31,128 species. The data are for seed plants only and include exotic species, but not commercially-grown crops.

[4]Compiled by Chapter 3 from Del Hoyo et al., 1992a, b; Gill and Donsker (2017); Rodewald (2015); Wetmore et al. (1957). World total data from Gill and Donsker (2017).

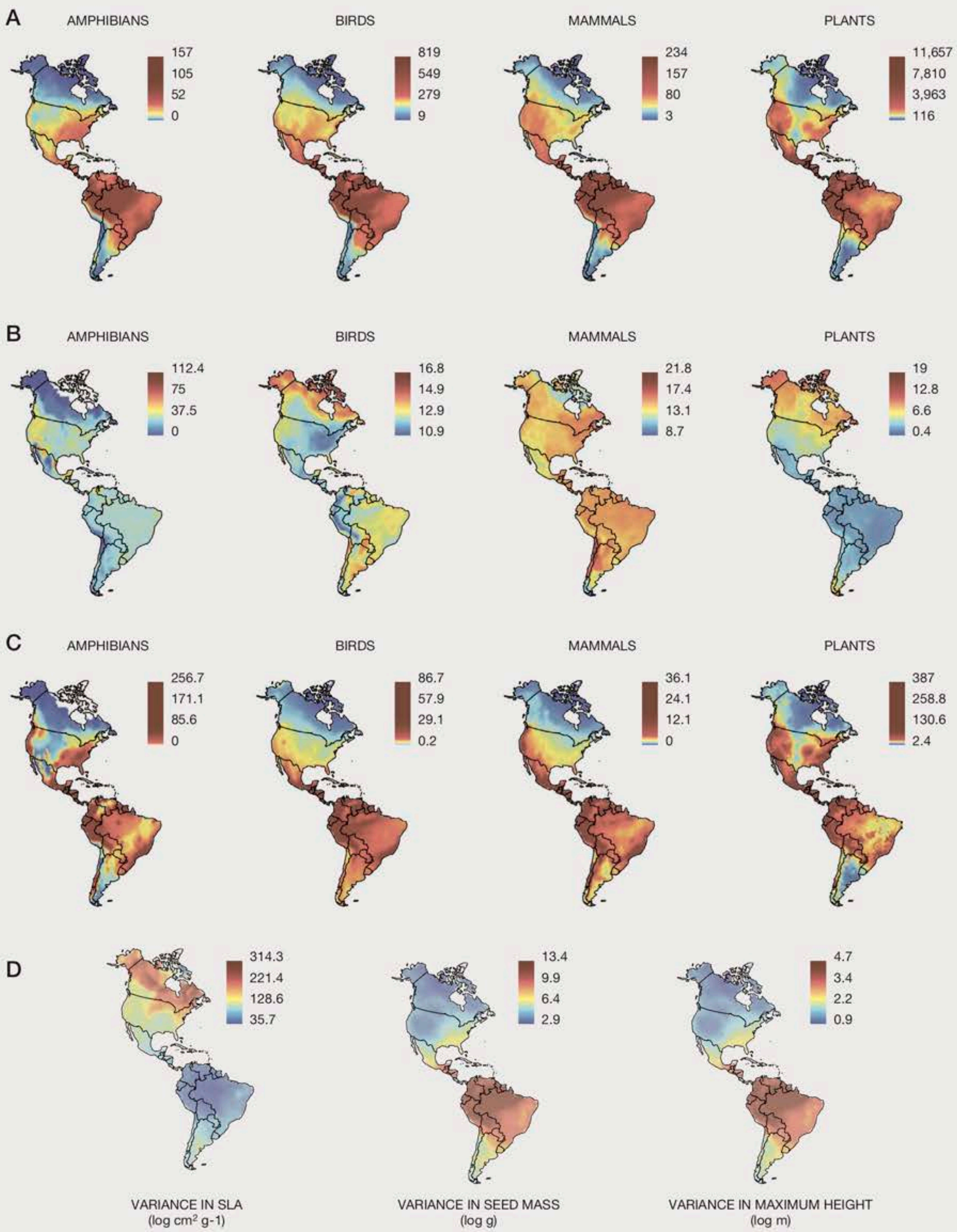
[5]North America – Bradley et al. (2014); Caribbean – Upham (2017) and IUCN, (2014); Mesoamerica – IUCN, (2014); South America – IUCN, (2014). Total for calculation of world %: IUCN Red List.

[6]amphibiaweb.org.

[7]reptile-database.org

[8]After the Summary for Policy Makers (SPM) for the Americas assessment was completed in early December 2017, in a paper published in *Science* on 14 December 2017, Ulloa Ulloa et al. (2017) reported 124,993 species of vascular plants (seed plants, ferns and fern allies) for the Americas region found in 6227 genera and 355 plant families. The number of species reported corresponds to 33% of the world total.

Figure 3 4 **Terrestrial biodiversity across the Americas in amphibians, birds, mammals and plants, reported as: A. species richness (SR); B. evolutionary distinctiveness (ED); C. phylogenetic endemism (PE); and D. plant functional diversity (FD). Source: own representation.**



FD was measured as the variance in specific leaf area (log cm²/g), seed mass (log g) and plant maximum height (log m) in 1 degree latitude and longitude grid cells using BIEN 2 and TRY Data. The red end of the color spectrum indicates greater SR, ED, PE and FD. Vertebrate metrics were calculated in 108x111 km cells; plant richness, ED, PE and FD in 100x100 km cells. A quantile color scale that emphasizes variation in lower values is used for species richness and PE. Species distributions: Birds, BirdLife International & NatureServe (2012); amphibians and mammals, IUCN, (2009); plants, Botanical Information and Ecology Network (BIEN 2) database, Enquist *et al.* (2016); Maitner *et al.* (2017). Phylogenies: Mammals, Fritz *et al.* (2009); birds, Jetz *et al.* (2012); amphibians, Pyron (2014); plants ED, BIEN 3 phylogeny, Maitner *et al.* (2017); plants PE, Zanne *et al.* (2013) (trimmed to genus). R software (R Development Core Team, 2017) and picante package (Kembel *et al.*, 2010); Nipperess and Wilson (2017) were used for calculations of the phylogenetic metrics and the R packages raster (Hijmans, 2016) and letsR (Vilela & Villalobos, 2015) were used to create the rasters.

Continent-level spatial patterns. The development of new biodiversity metrics that go beyond traditional species richness and better spatial data coverage of species over the past 15 years have greatly improved our understanding of how biodiversity is distributed at a finer geographical scale within the Americas. New patterns have emerged that are highly relevant for the valuation of biodiversity across the region. See the glossary for definitions of the biodiversity metrics assessed.

Reflecting the subregion-level survey data (Table 3.1), amphibians, birds, mammals, and plants all show high species richness in tropical South America and Mesoamerica (Figure 3.4a). For mammals and amphibians, the highest richness is found in the Andes, the coastal northwest of South America and the Atlantic coast of Brazil; plants (Figure 3.4a) reach their highest richness in Mesoamerica, the Andes and other regions of South America. Avian richness shows peaks in both the lowlands and parts of the Andes. Evidence from ants (Dunn et al., 2009) and soil fungal communities (Tedersoo et al., 2014) suggests that these taxa may also reach their peak diversity in tropical regions, although considerable gaps exist in spatial sampling for these groups. Outside of the tropics, amphibians and plants show moderately high species richness in the southeastern USA (Figure 3.4. a) (Buckley & Jetz, 2007), and plants and mammals both reach high or moderately high richness in the western USA.

In contrast to species richness, which is broadly congruent across taxa and reaches its peak in tropical South America, highest evolutionary distinctiveness is found outside of the tropics for all taxa (Figure 3.4. b). For the taxa where information is available, this indicates that the regions where co-occurring species are more distantly related on average tend not to be found in the tropics. Among amphibians, high evolutionary distinctiveness is found in western North America and parts of Mesoamerica. Mammals have high evolutionary distinctiveness throughout the Americas, especially in the Mediterranean region of southwestern South America. Birds and plants both have hotspots of evolutionary distinctiveness at high latitudes, indicating that even in these regions where low numbers of species persist, the species that do occur are drawn from distinct branches across the tree of life. Birds also achieve moderately high evolutionary distinctiveness in the tropical lowlands of South America. Overall, these trends indicate that subtropical, temperate or boreal regions can be rich in certain dimensions of biodiversity. This, of course, does not mean that the tropics have less overall evolutionary diversity, but rather that tropical species often co-occur with many close relatives, reducing their evolutionary distinctiveness.

In all taxa, high phylogenetic endemism occurs in Mesoamerica and in parts of tropical South America, particularly the coastal northwest and tropical Andes (Figure 3.4. c). Amphibians, mammals, and plants have further hotspots of phylogenetic endemism in the western USA, and amphibians and plants also have high phylogenetic endemism in the southeastern USA. Central and part of southern Chile also stand out for some groups. With some deviations, geographic patterns of phylogenetic endemism in these particular areas of the Americas tend to mirror species richness, signifying overall that they generally house large numbers of evolutionary distinct species and lineages not found elsewhere. Such areas are worthy of special concern in conservation decision-making but are sometimes located where protection measures are still poor (3.5.2).

Variation in functional traits, a measure of functional diversity, can tell us about the diversity of ecological adaptations among a set of organisms and the potential of particular ecosystems to adjust to environmental change. Data are available on three functional traits for plants; specific leaf area, seed mass and plant height. Specific leaf area (the area of a leaf divided by its dry weight) is tightly linked to photosynthetic rates and nutrient content. It is indicative of the life history strategy of the plant along a spectrum ranging from rapid growth and competitive resource capture to slow growth and stress tolerance (Wright et al., 2004). Seed mass is indicative of reproductive and dispersal strategy (Leishman et al., 1995; Moles et al., 2005), and plant height is a critical indicator of life history, indicating growth form and habit (Loehle, 2000). These three traits are important for understanding major axes of variation in plant function and ecological strategy (Westoby, 1998). As with species richness, we tend to see the greatest diversity in seed mass and height of vascular plants in tropical regions of the Americas (Lamanna et al., 2014) (Figure 3.4. d). Nevertheless, temperate regions tend to be enriched in functional diversity for specific leaf area relative to tropical areas (Figure 3.4. d). This might reflect a tendency to retain more diversity in leaf economic strategies under harsher and less equitable climatic conditions (Lamanna et al., 2014; Swenson et al., 2012). Variation in different plant functional traits is maximized in different regions. Likewise, different components of diversity are highest in different regions and these patterns vary by taxonomic group. As a consequence, conservation efforts across regions will be crucial for maintaining both the diversity of ecological strategies we observe in plants and the full spectrum of biodiversity across the tree of life, the basis of many NCP.

3.2.2.3 Patterns and trends in alien and invasive alien species

Status. We define alien species as species that become distributed beyond their native ranges intentionally or unintentionally aided by humans. The introduction and spread of alien species in the Anthropocene has led to greatly heightened levels of dispersal of organisms around the globe. Invasive alien species are alien species that modify ecosystems, causing potential damage to the environment, human health, and consequently, the economy. The distinction between these two categories is not always clear because designating an alien species as an invasive species requires detailed studies and objective and comparable criteria. The economic damage caused by alien invasive species can be severe. For example, globally, invasive insects (some of which carry diseases) are estimated to cost a minimum of $70.0 billion per year, while associated health costs exceed US$6.9 billion (Bradshaw et al., 2016). Control of invasive species requires knowledge of global and local introduction trends and distinguishing harmful alien species from the more benign ones; that said, not all alien species are harmful (Table 3.2).

Comprehensive data on naturalized alien species for the Americas is available for plants and birds. Currently the North American (which includes Greenland and Mexico) and the South American (which includes Mesoamerica south of Mexico and the Caribbean) biogeographic regions are home to 3,513 (39%) and 1,806 (20%) respectively of the world’s 9004 plant species that have been introduced from one continent to another (Van Kleunen et al., 2015, and personal communication). Additional intra-continental plant movements beyond their natural ranges within North and South America, bring the total numbers of alien records to 5,958 and 3,117, respectively. North America has been a much larger donor of alien plant species to other continents than has South America; additionally North America, as defined by IPBES, is one of the most heavily invaded areas of the world (Van Kleunen et al., 2015). The Caribbean subregion is also strongly invaded in relation to its land area (see also Figure 3.5, where there are many plant species).

Some 3,661 alien bird introductions (first known occurrence of a given species in a given country) were reported across the globe from 1500 to 2000 (Dyer et al., 2017). Relative to other regions the Americas, particularly the North American and Caribbean subregions, support large numbers of alien birds (Dyer et al., 2017). Reports of introduced birds are currently lacking in some tropical areas in northern South America (Dyer et al., 2017).

Table 3.2. Multiple effects of mostly recent terrestrial alien introductions in the Americas. Alien species can have both negative and positive impacts on humans and biodiversity. See chapter 4 for additional examples. See chapter 4 for additional examples. 🔴 = negative impact; 🟢 = positive impact.

Table 3 2 **Multiple effects of mostly recent terrestrial alien introductions in the Americas. Alien species can have both negative and positive impacts on humans and biodiversity. See Chapter 4 for additional examples. 🔴 = negative impact; 🟢 = positive impact.**

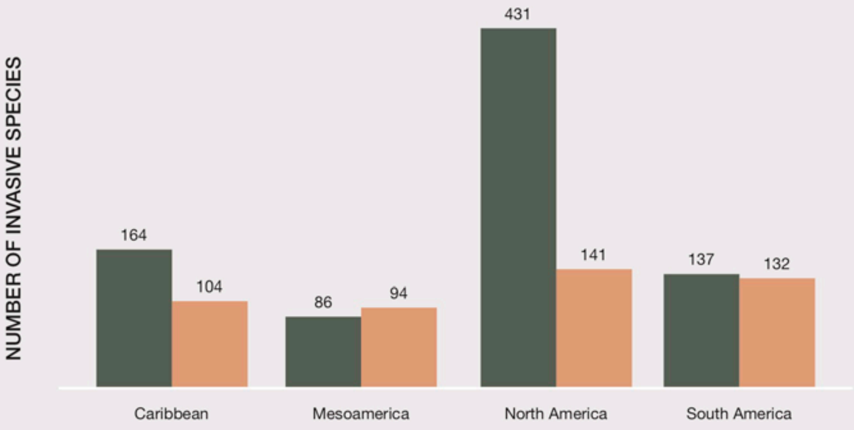
Sources: 1 Morales *et al.* (2013); Aizen *et al.* (2014); 2 Sanguinetti & Singer (2014); 3 Dangles *et al.* (2008); 4 Herms & McCullough (2014); 5 Martyniuk *et al.* (2015); 6 Peña *et al.* (2008); Taylor *et al.* (2016); 7 Zamora Nasca *et al.* (2014); 8 García *et al.* (2015); 9 Baruch & Nozawa (2014); 10 Svriz *et al.* (2013); 11 Pauchard *et al.* (2009); Barros & Pickering (2014); 12 Muñoz & Cavieres (2008); 13 León & Vargas-Ríos (2011); 14 Díaz-Betancourt *et al.* (1999); 15 Rodrigues da Silva & Matos (2006); 16 Choi (2008); 17 Jiménez *et al.* (2014).

	Insects
🔴	European *Bombus terrestris* reduces fitness of native plants and replaces the native bumblebee, *B. dahlbomii*.[1]
🟢	Introduced bees increase fitness in some native orchids.[2]
🔴	Three potato moths reduce crop harvest in the northern Andes.[3]
🔴	The Asian emerald ash borer beetle (*Agrilus planipennis*) has killed millions of ash trees in N. America.[4]
	Plants
🔴	Seed set on the native *Austrocedrus chilensis* is reduced by interference of introduced conifer pollen.[5]
🔴	Encroachment of exotic plantation tree species into native forests reduces habitat area.[6]
🔴	*Ligustrum lucidum* reduces soil water availability in secondary forests.[7]
🔴	*Teline monspessulana* increases fire proneness in native forests.[8]
🔴	Aggressive *Syzygium jambos* interferes with natural regeneration in abandoned coffee plantations.[9]
🟢	*Rubus rubiginosa* acts as a nurse plant for regeneration of native forest trees on drier sites.[10]
🔴	Non-native species on trails homogenize the floras of protected areas, reducing landscape value.[11]
🔴	*Taraxacum officinale* reduces pollinator visits on native species in the high Andes of central Chile.[12]
🔴	*Ulex europaeus* invades páramos, displacing native species and possibly harming water supply.[13]
🟢	Introduced weeds in the Americas include many edible species.[14]
🔴	Post-fire invasion by *Pteridium aquilinum* in the Atlantic rainforest hinders natural forest regeneration.[15]
	Mammals
🔴	North American beaver affects forest hydrology and forest regeneration in Tierra del Fuego.[16]
🔴	American Mink preys on the eggs of water birds and the iconic Magellanic woodpecker.[17]

Although much progress has been made, we currently cannot say how many alien species in the Americas are harmful. Comprehensive risk analyses are lacking in most countries. In general, the number of harmful species is likely to higher than currently visualized because detailed studies are lacking and due to the fact that many potentially strongly invasive species will be still in a lag phase. In Mexico, a comprehensive risk analysis found 41% of 472 species (including aquatic species) analyzed out of a total of 1,683 potentially invasive species to be very high-risk species (Gonzalez Martínez et al., 2017).

Figure 3 5 **Invasive alien plant and animal species considered to threaten native biodiversity and ecosystems listed in the Global Invasive Species Database (GISD) that are found in the four subregions of Americas.**

Data include a few marine and freshwater species. Grey bars are species that have been reported somewhere in that subregion as being strongly invasive; orange bars are additional species listed in GISD that occur in the subregion but that are not necessarily invasive there or whose invasive status is unknown. Source: Data from Global Invasive Species Database http://www.iucngisd.org/gisd/. Accessed March, 26 2017.

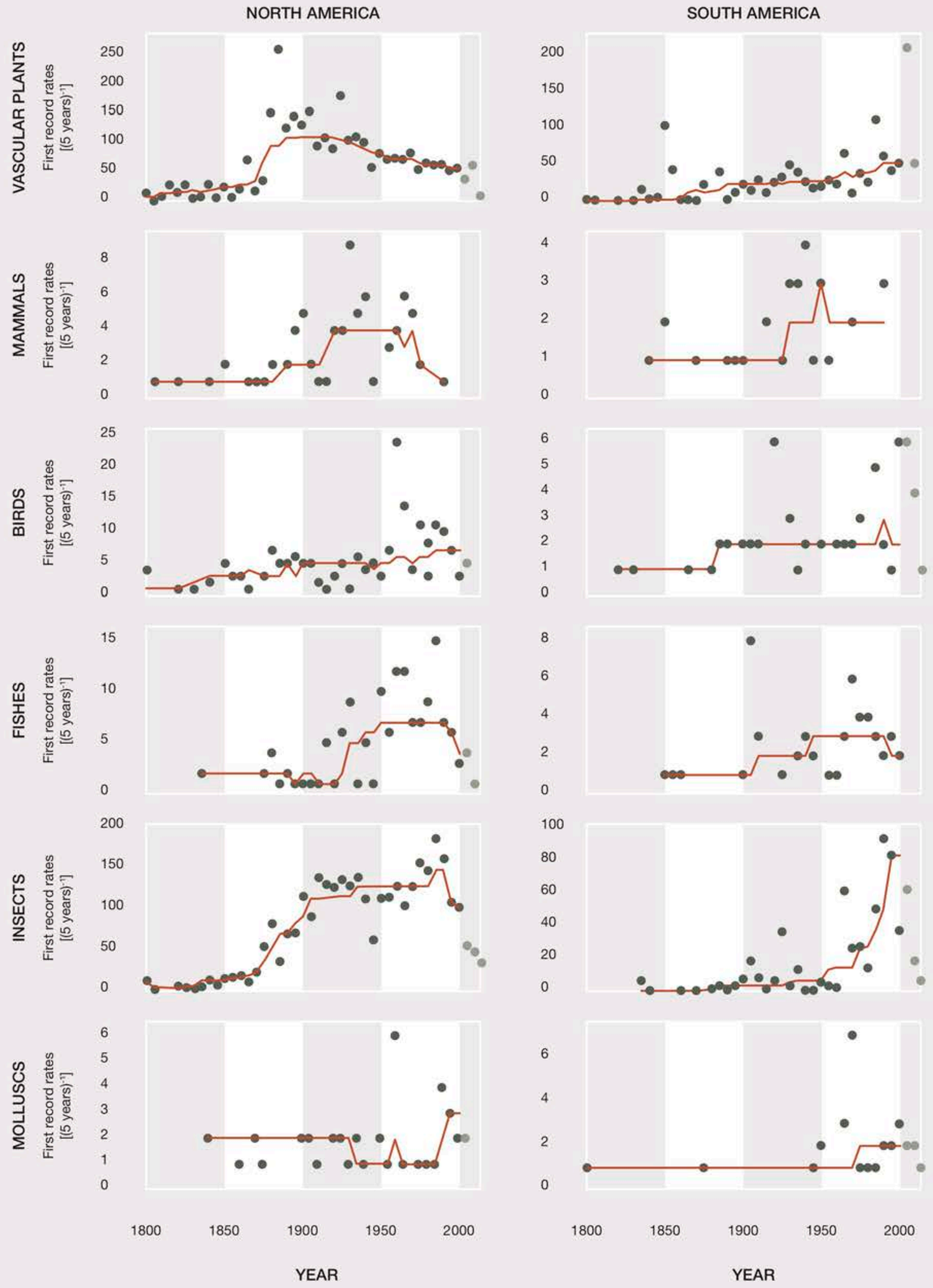


Across all taxonomic groups, some 521 species considered to be harmful to biodiversity in Global Invasive Species Database are known to be strongly invasive somewhere in the Americas. North America has far more such species than the other subregions, but for its small land area, the Caribbean is clearly very susceptible to invasion **(**Figure 3.5). Additional species found in Global Invasive Species Database that are not considered to be invasive at the moment in a particular subregion could eventually become invasive (Figure 3.5). For the World's 100 Worst Invaders found in Global Invasive Species Database, 78% have been recorded to occur in at least one subregion of the Americas. Beyond invasive species that harm biodiversity, many alien species have negative impacts on agriculture and forestry. For example, in Brazil, more than 500 species of alien pathogenic fungi, 100 viruses, 25 nematodes and one protozoan attack crops and reduce crop production an estimated 15% (Pimentel, 2002). Chapter 4 provides more information on the effects of harmful invasive species and on their drivers.

Recent trends. Globally, 37% of all recorded naturalized aliens from a wide spectrum of taxonomic groups were recorded for the first time as recently as 1970–2014 (Seebens et al., 2017). This signifies that invasion risk is currently high and with increasing globalization will not cease. For the Americas, rates of appearance for different groups have varied over time, with a tendency for steeper early climbs and an earlier tendency to decline in North America than in South America (Figure 3.6). Insects showed a very rapid rate of increase in South America as of the 1950s.

Figure 3 6 **Trends in the appearance of alien species in north america and south america from 1800 to 2000.**

Source: Based on data in the global alien species first record database www.dx.doi.org/10.12761/SGN.2016.01.022. Accessed: August 25, 2016.



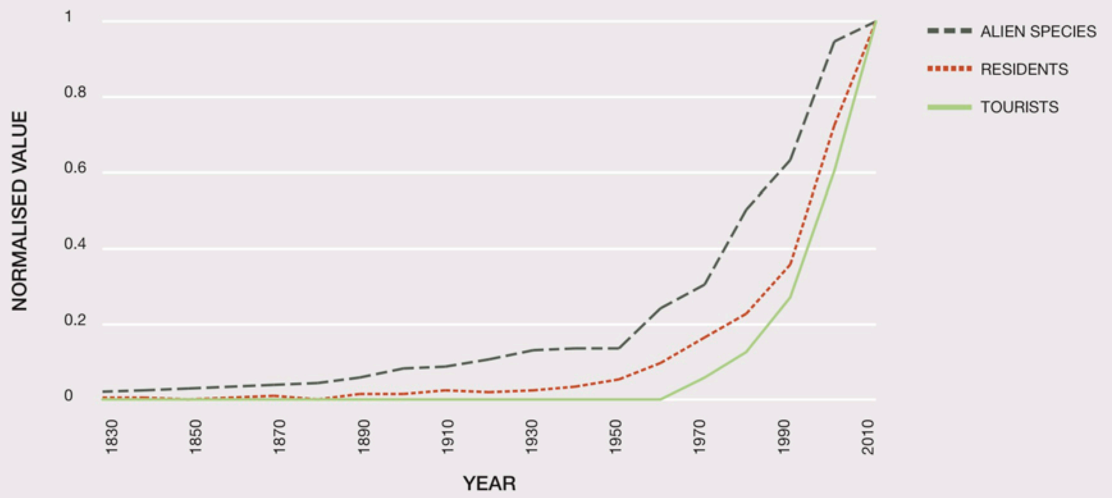
For birds, half of the naturalized alien introductions worldwide occurred after 1956, in concert with increasing globalization and economic growth. As with plants, early bird introductions came mostly from

Europe. However, more recently the Indian subcontinent, Indochina, and sub-Saharan Africa have become important sources of alien birds (Dyer et al., 2017). For the Americas, as of 1983, at the country level, 102 new alien birds were registered for the Caribbean subregion, 8 the Mesoamerican subregion, and 19 for South America. At the individual state (USA) or province (Canada) level, 163 were recorded for the North America subregion - calculated from Supplementary material (Dyer et al., 2017).

Box 3.1. Alien species in the Galápagos Islands.

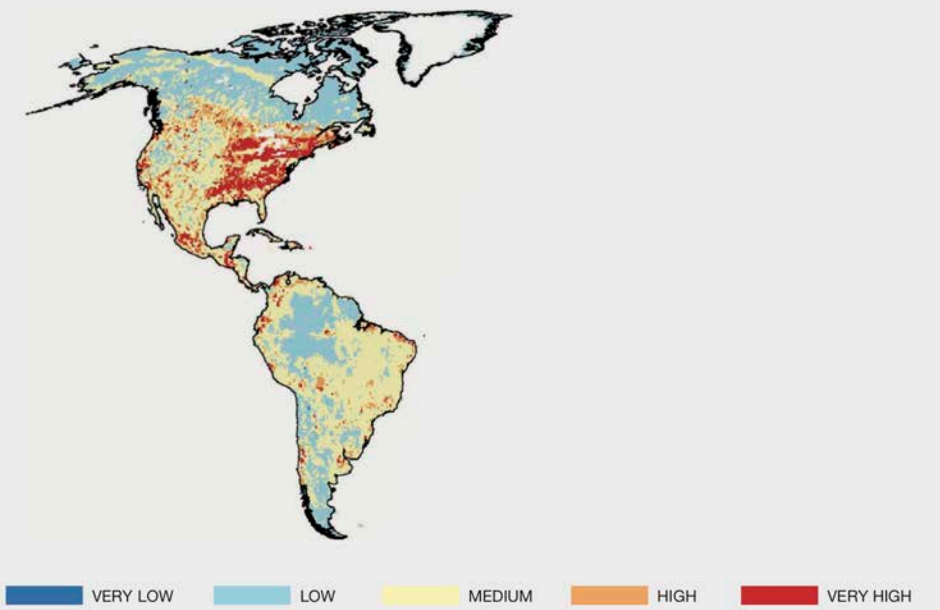
Described by UNESCO (United Nations Educational, Scientific and Cultural Organization) as a “living museum and showcase of evolution”, the Galápagos Islands, a World Heritage site, are today a major tourist attraction. Some 1,476 of 1,579 alien terrestrial (and marine) species have become established on the islands (Toral-Granda et al., 2017); 50% of aliens were first reported after the 1990s and just over 50% were introduced through unintentional human assistance. The rate of introduction represents an average of around three species per year. Geographic origins and modes of introduction have diversified over time, reflecting the increase in human influence on the islands. In general, islands are prone to invasions. The mythical oceanic Robinson Crusoe islands are also strongly invaded (Wester, 1991).

Figure 3 7 **Normalised decadal values for the cumulative number of alien species, residents and tourists, in the Galápagos Islands, ecuador. Source: Based on data given in supplementary material in Toral-Granda *et al.* (2017).**



Overall, alien introductions and their spread are likely to continue in the Americas (Seebens et al., 2017) opening the door to additional negative effects on biodiversity, forestry and agriculture. Modeling suggests that many established alien species in the Americas do not yet fully occupy their climatic niches (Arriaga et al., 2004; Peña-Gómez et al., 2014) and thus can be expected to expand further, facilitated by disturbance. We are currently in a modern era of assisted dispersal heightened by global travel, tourism, and the introduction of pets and pest-carrying plant parts (see Chapter 4). A dramatic example of how alien species have increased recently in relation to increasing vector availability is seen in the Galápagos Islands (Box 3.1, Figure 3.7).

Figure 3 8 Invasion threat across the Americas in the 21st century. Source: Modified from Early *et al.* (2016).



Knowing which geographic areas are likely to receive more alien species is useful for the development of early-warning systems. According to a recent analysis of current invasion vectors and environmental susceptibility to invasion, the threat of invasion is very unevenly distributed across the Americas (Early et al., 2016) (Figure 3.8). The dominant invasion vectors differ between high-income countries (imports, particularly of plants and pets) and low-income countries (air travel). Climate change, further biome transformation (e.g. 3.4.1.6) and increased fire frequency (e.g. 3.4.1.4) are expected to hasten the spread of invasive alien species once established. Given that strongly invasive alien species, in addition to signifying economic costs, have been the cause of many extinctions (Bellard et al., 2016), alien species are a component of biodiversity that requires attention.

3.2.3 Status and trends of freshwater biodiversity

3.2.3.1 Patterns of diversity for taxonomic groups

Taxonomic groups. The Americas hold the most diversified freshwater fish fauna in the world, with 1,213 species in the North American biogeographical region and 4,035 in the South American biogeographical region for a world total of over 13,600 species (Burkhead, 2012). Other freshwater taxonomic groups of note include crayfishes, with high diversity in the southeastern USA (Crandall & Buhay, 2008); amphibians, with nearly half of all salamander species found in North America; 40% of all water-dependent frog species found in the Neotropical realm (Vences & Köhler, 2008); 11 of the world's 23 crocodilian species (Martin, 2008); the vast majority of the world's temperate freshwater turtle species in North America (Bour, 2008); the most diverse freshwater bivalve fauna globally also in North America (Bogan, 2008); and an especially diverse assemblage of decapods in Central America (Wehrtmann et al., 2016).

Freshwater species contribute NCP in numerous ways. Freshwater mussels cleanse water (Nobles & Zhang, 2011). Fish regulate nutrients in water (Holmlund & Hammer, 1999). North American Pacific salmon transfer nutrients from marine to freshwater realms when they die en masse after migrating upstream (Flecker et al., 2010). In the Amazon, Orinoco, and parts of Central America, frugivorous fish disperse seeds for floodplain forest trees (Flecker et al., 2010). An estimated 450,000 tons of riverine fish are landed each year in the Amazon, with important implications for the food security of local people (Junk et al., 2007; McIntyre et al., 2016). However, riverine fish catch is estimated to be low in large North American rivers like the Mississippi, where recreational fisheries dominate commercial or artisanal fisheries (McIntyre et al., 2016). Overall, reported inland fish catch in the Americas is low compared to other regions (Bennett & Thorpe, 2008).

Status. Much freshwater biodiversity in the Americas is threatened, derived largely from catchment land use, water use and direct habitat alterations (Vörösmarty et al., 2010) (see Chapter 4 for discussion of drivers). Some 23% for the Nearctic and 22% for the Neotropics of freshwater mammals, amphibians, reptiles, fishes, crabs and crayfish collectively fall here. The well-studied North American biogeographical region freshwater fish fauna in the 20th century had the highest extinction rate worldwide among vertebrates (Burkhead, 2012). Some 72% of freshwater mussels in the USA and Canada were considered imperiled as of the early 1990s (Williams et al., 1993). In Central America, 42% of ca. 500 known amphibian species have been assessed as threatened, with stream-dependent species at particular risk (Whitfield et al., 2016). Regions with low threat are remote areas in northern Canada, Alaska, and the Amazon.

Recent trends. In North America (including Mexico), since 1989, the number of threatened freshwater fishes has increased by 25%; extinctions peaked after 1950 with 7.5 extinct taxa per decade post-1950 (Burkhead, 2012). This level of extinction gives reason for great concern. In the Caribbean native fish species continue to decline and be extirpated with dam building, pollution and overharvesting exerting considerable pressure (Cooney & Kwak, 2010). In the 1970s, a noticeable decline in populations of freshwater turtles in the Amazon was observed (Eisemberg et al., 2016). Amphibian population declines in Mesoamerica and South America have been documented largely beginning in the 1970s–1990s, with the majority in the 1980s (Young et al., 2001). Freshwater mussel extinctions have been documented in the USA from the beginning of the 20th century, with a peak of eight extinctions in the 1920s through the 1940s and seven documented extinctions in the 1970s (Haag, 2009).

3.2.3.2 Patterns and trends in alien and invasive species

Status and recent trends. Data on freshwater alien species is scattered, making it difficult to provide an overall picture for the Americas and its subregions. Where databases are available, numbers of alien species can be high, as seen in over 1000 species in the USA (plants excluded) (Fuller & Neilson, 2015) and 50 species of fishes (including marine species) in Mexico (Mendoza & Koleff, 2014) (see also Box 3.2). The impacts of aquatic alien species are multiple and can be severe (Table 3.3). The spread of some aquatic invasive alien species, moreover, has been very rapid, leaving cause for concern.

In North America, alien freshwater species have been arriving for close to two centuries and continue to arrive. Some of the earliest known introductions occurred in the late 1800s when fish were transported from coast to coast (Benson & Boydstun, 1999). Crayfish and other freshwater organisms were moved from the southeastern USA to the western USA to serve as game species or forage for game species. Temperate piscivorous and carnivorous fish species have been reported to cause much harm to native fish fauna, especially in Cuban freshwaters, Lake Atitlán (Guatemala) and Lake Titicaca (Bolivia and Perú) (Revenga & Kura, 2003).

The zebra mussel (*Dreissena polymorpha),* native to Europe, and the Asian clam *(Corbicula fluminea*) are estimated to cost the $1 billion a year, largely through impacts to infrastructure (Pimentel et al., 2005). Their spread has been recent, with the first established zebra mussel population recorded in the USA in 1988 (Benson, 2012). Some freshwater invasive species in South America have also spread very rapidly. For example, the exotic freshwater water-fouling mussel, *Limnoperna fortunei*, was introduced into Río de la Plata estuary in 1991; from there it spread at a rate of up to 250 km.year^{-1} and is now found in freshwater systems in Argentina, Uruguay, Paraguay, Brazil, Bolivia (Darrigran et al., 2012; Darrigran & Ezcurra de Drago, 2000; Oliveira et al., 2015). This mussel, which is similar to invasive Dreissinids in North America, has altered benthic communities and is predicted to expand further. This example shows that insufficient measures to prevent the introduction of invasive aquatic species can have severe consequences. The most invaded freshwater system in the Americas, and a warning to what can happen without adequate control from the beginning, are the Great Lakes of North America (Box 3.2, Figure 3.9). Other examples of freshwater invasions are shown in Table 3.3.

Table 3.3. Multiple effects of freshwater invasive species in the Americas. See Chapter 4 for additional examples. for additional examples. . • = negative impact; • = positive impact; ? • = unsure negative impact.

Table 3 3 Multiple effects of freshwater invasive species in the Americas. See Chapter 4 for additional examples. • = negative impact; • = positive impact.

Sources: 1 Junk (2007); 2 Thompson *et al.* (1987); 3 Brown & Maceina (2002); 4 Perry *et al.* (2001); 5 Pyron *et al.* (2017); 6 Wilson *et al.* (2011); 7 Howard (2016); 8 Leal-Flórez (2008); 9 Montecino *et al.* (2014); 10 Villamagna & Murphy (2010); 11 Bacheler *et al.* (2004).

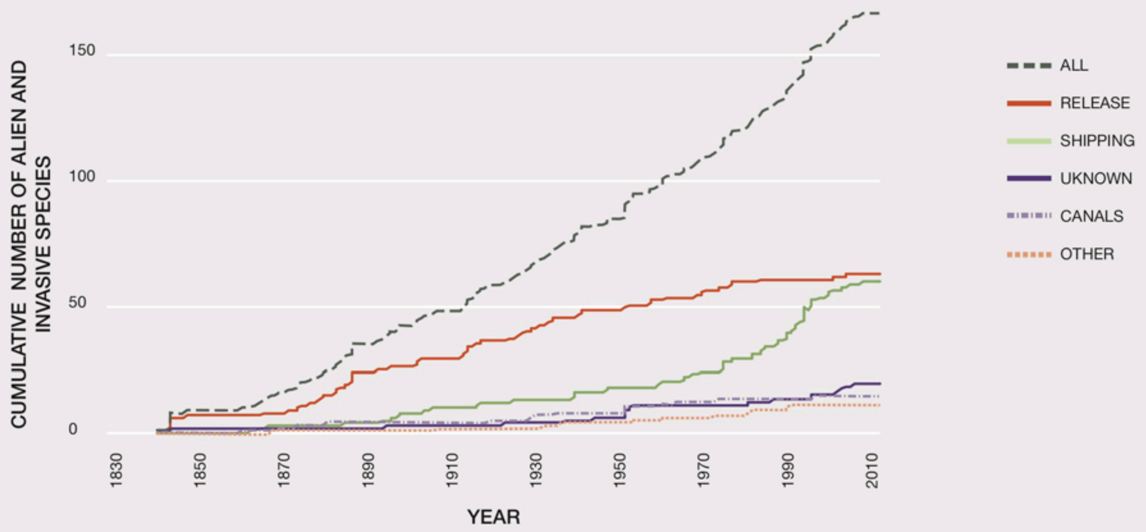
	Invasive species
•	Introduction of rainbow trout in Lake Titicaca decreased native fish food supply.[1]
•	Purple loosestrife (*Lythrum salicaria*) has reduced the biomass of 44 native plants and dependent endangered wildlife species.[2]
•	Infestations of the aquatic weed hydrilla *(Hydrilla verticillata)* have reduced angling up to 85%.[3]
•	The rusty crayfish (*Orconectes rusticus*), native to the Ohio River basin, is spreading in the USA and replacing native species.[4]
•	Asian carp contributed to modifications in native fish assemblages in the Wabash River, USA, likely by competing with native planktivore / detritivore fishes.[5]
•	The cane toad (*Bufo marinus*) has spread to the Caribbean and is killing the threatened endemic Jamaican boa *(Epicrates subflavus)*.[6]
• ?	Hippos introduced into Colombia are now multiplying and may contribute to eutrophication via their waste.[7]
• •	Accidental introduction of *Oreochromis niloticus* into Colombia's Santa Marta estuary has provided local fishermen with a source of income during short periods of low salinity, when native fish catches drop. However, this same species has had negative impacts in many other American ecosystems.[8]
•	The alien diatom *Didymosphenia geminata* recently expanded in southern Chile and Argentina greatly reducing aesthetic value of lakes and streams.[9]
• •	The water hyacinth (*Eichhornia crassipes*), native of lowland tropical America, has become invasive in many countries of the region with mostly negative effects on waterways, but some positive effects on biodiversity.[10]
•	In Puerto Rico, there is a significant overlap in diet between the native *Gobiomorus dormitor* and largemouth bass *Micropterus salmoides* introduced from North America.[11]

Box 3.2. The Great Lakes history with invasive species.

The Great Lakes in North America have accumulated an excess of 165 alien species in what is still an ongoing process (Figure 3.9). Some species have had significant negative impacts on aquatic ecosystems (Higgins & Vander Zanden, 2010). Among the most damaging is the sea lamprey (*Petromyzon marinus*), which appeared in the 1830s and spread throughout the Great Lakes during the 20th century, impacting several fisheries. Zebra mussels and quagga mussels, first detected in the late 1980s, create dense colonies that harm ecosystems, harbors and waterways and clog water intakes in water treatment facilities and power plants.

Figure 3 9 **Trends in the accumulation of alien and invasive species in the North American Great Lakes over time.**

The upper line of the graph shows total cumulative number; the other lines show the contribution from various vectors. "Release" includes both intentional and unintentional; "other" includes railroads, highways, aquaria, and baitfish. Source: Data compiled from Kelly (2007), Kelly *et al.* (2009) and Ricciardi (2006).



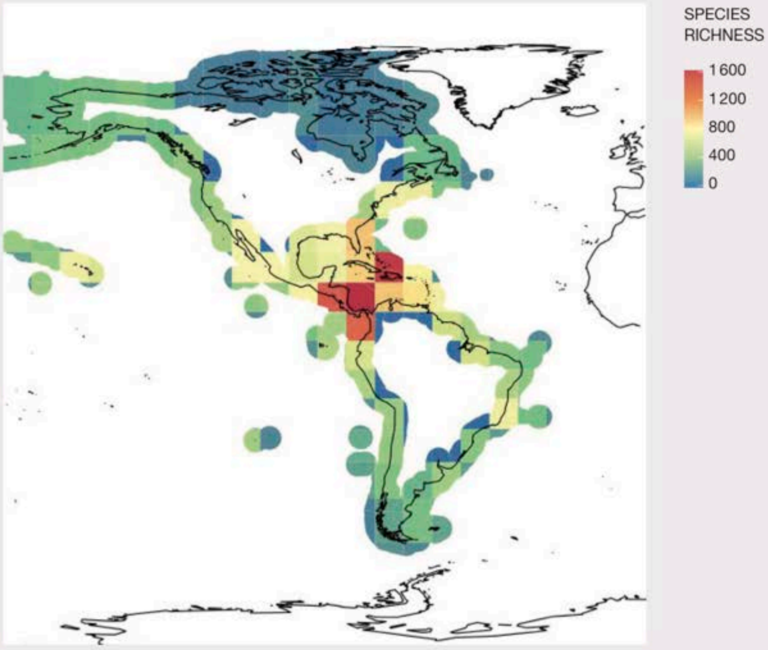
3.2.4 Marine biodiversity

3.2.4.1 Patterns of diversity for taxonomic groups

Status. Marine life in the Americas is found in the Atlantic, Pacific and Arctic oceans, and in the Caribbean Sea. Atlantic and Pacific offshore and deepwater areas (> 200 m) encompass a range of habitats with a wide diversity of species (OBIS, n.d.) (Figure 3.10). Exceptional diversity is being revealed for the oceans. Including all taxonomic groups (except bacteria and phytoplankton), 12,046 marine species have been found in the Caribbean realm (Miloslavich et al., 2010), 10,201 in the Humboldt Current System, 6,714 in the Tropical East Pacific, 9,103 on the Brazilian shelves, 2,743 in the Tropical West Atlantic, and 3,776 on the Patagonian shelf in South America (Miloslavich et al., 2011). These numbers are considered to be conservative (see 3.6). Marine mammals in the Americas include 74 cetacean species, 22 pinnipeds, 3 sirenians, 3 mustelids and the polar bear. Additionally, six of the world's seven sea turtles and more than 400 chondrichthyan species occur in the Americas. The Arctic Ocean, in its waters, ice and seafloor, hosts unique biodiversity of many thousands of species, including mammals, seabirds, fish, invertebrates, and algae (Gradinger et al., 2010) in a rapidly changing environment (see Chapter 4). The Caribbean basin deep-sea species database (OBIS. n.d.) lists 1,530 species from 12 phyla, but much more work is needed (Miloslavich et al., 2010). The Caribbean Sea holds most of the Americas' biodiversity associated with coral reefs.

Figure 3 10 **Species richness across coastal fishes, marine mammals, mangroves, corals, foraminiferans, euphausids, cephalopods, tuna and sharks in the coastal ecoregions of the Americas.**

Source: own representation from supplementary data in Tittensor *et al.* (2010).

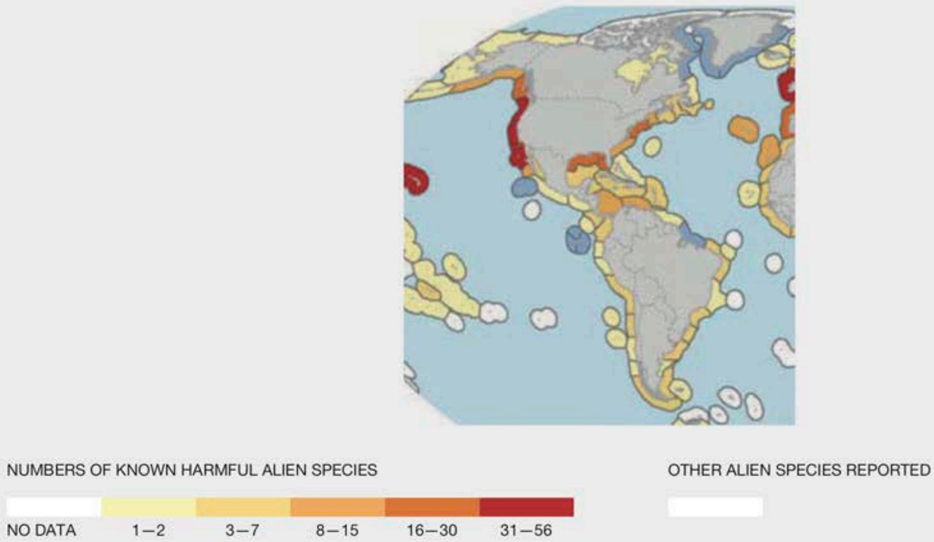


In many species of coastal fish, mangroves, seagrasses, squids, non-oceanic shark species, and corals, diversity generally peaks near the equator (Tittensor et al., 2010). In contrast, pinniped (seals and sea lions) diversity is highest in polar regions. Cetacean species diversity peaks in the subtropics in both oceans, and is highest on the Atlantic coast of Argentina (Tittensor et al., 2010). Shark species peak in biodiversity between 30 and 40 degrees N and S; southeastern Brazil and the southeastern USA are considered global hotspots of shark biodiversity with high species richness, functional diversity, and endemism (Lucifora et al., 2011). Brazil alone has 31 endemic shark species (Lucifora et al., 2011). Seaweed biodiversity peaks in temperate regions around 35 degrees latitude N and S in the Pacific (Gaines & Lubchenco, 1982), although it is also highly diverse in the Caribbean (Kerswell, 2006). Kelp diversity is greatest in colder parts of both oceans, and algal diversity reaches its nadir in the southeastern Atlantic (Argentina, <100 species) (Kerswell, 2006). The America's host hundreds of thousands – if not millions – of invertebrate species; their biogeographic patterns are still poorly known (Sala & Knowlton, 2006). Invertebrate diversity within many distinct taxonomic groups generally (with some exceptions) follows the latitudinal trend of increasing species diversity per area at lower latitudes, as seen in South American crabs on both coasts (Astorga et al., 2003), and fish (Rohde et al., 1993), molluscs (Roy et al., 1998) and foraminifera (Rutherford et al., 1999) in North America. Different biogeographic regions, reflecting major oceanographic features, have distinct invertebrate species assemblages off South America, North America, the Arctic and the Caribbean. This pattern is exemplified by the spatial distribution of the estimated 1,539 species of echinoderms inhabiting Latin America (Pérez-Ruzafa et al., 2013). The Western Atlantic and the coast of South America host an exceptionally high diversity of the world's 2064 ophiuroid echinoderms (335 species) with high rates of endemism (Stöhr et al., 2012).

3.2.4.2 Patterns and trends in marine invasive species

Status. Pagad et al. (2017) document 2,103 introduced marine species worldwide, of which 305 are considered strongly invasive. According to this source, the North American continent has 388 alien marine species (70 invasives); Mexico, 94 (6 invasives); the Caribbean Sea, 47 (5 invasives); Brazil, 49 (9 invasives); Argentina, 25 (1 invasive) and Greenland, 1 (not invasive).

Figure 3 11 **Relative invasion levels in marine habitats across the Americas based on the number of species with high ecological impact scores per ecoregion. Source: Modified from Molnar *et al.* (2008).**



In general, North American waters are more heavily invaded than those of other subregions (Figure 3.11). However, differences between North and South America are less evident when considering invasive algae (including native invasive species) (Figure 1.A in Seebens et al., 2016). Miloslavich et al. (2011) report more alien species at cooler latitudes in South America, but this difference might be influenced by sampling density at other latitudes. Most alien and invasive invertebrate and algal species are found in bays and estuaries, with few occurring on outer coasts (Ruiz et al., 2015). San Francisco Bay, USA, may the most invaded marine region on Earth, with more than half its fish and most of its benthic invertebrates being non-native (Cohen & Carlton, 1998).

As in terrestrial habitats, recent invasions may not be detected for many years. Controlling the introduction of marine species and their impacts, however, is far more difficult than controlling terrestrial and freshwater species given that they are not so obvious. Moreover, introduced marine species can transport many other alien species. For example, the American oyster, *Crassostrea virginica*, was introduced to the Pacific coast to supplement stocks of local species. Oyster drills (*Urosalpinx cinerea*), slipper shells (*Crepidula fornicata* and *C. plana*), polychaetes (*Polydora cornuta*), and cordgrass (*Spartina alterniflora*) may have been introduced with them (Ray, 2005). Worms used for live bait (*Glycera dibranchiata*) are shipped packed in seaweed, which carries many potentially invasive organisms such as snails, crabs, isopods, insects, plants and algae (MD Sea Grant, n.d.). Some species arrive by multiple mechanisms, e.g. the Chinese mitten crab (*Eriocheir sinensis*) may have arrived in ballast water and in live trade as food (Ruiz et al., 2000) or as pets from the aquarium trade (Dee et al., 2014; Smith et al., 2008). Overall strategies to deal with marine invasion require international collaboration.

Recent trends. As is occurring in terrestrial and freshwater systems, the spread of alien species in marine systems of the Americas continues. Marine species once established, can spread very rapidly. The Asian green mussel *Perna viridis*, native to the Indo-Pacific, was first observed in Caribbean waters in 1990 (Agard et al., 1992). Within 10 years, green mussels were found along the coasts of Venezuela, Jamaica and Tampa Bay, Florida (Benson et al., 2001; Buddo et al., 2003; Ingrao et al., 2001; Rylander et al., 1996). Rates of marine introduction seem to be increasing in some places. For example, Cohen and Carlton (1998) estimated that the San Francisco Bay and Delta ecosystem has received about one new invasive species every 36 weeks since 1850: as of 1970, the rate increased to one new species every 24 weeks. A huge number of marine species (280) were recently found to have crossed the Pacific from Japan to the west coast of North America on debris swept to sea by the 2011 tsunami (Carlton et al., 2017), warning that increasing amount of debris in the oceans are a potential source of invasive marine species. Invasions related to human food production are a current concern. Non-native shrimp (Asian tiger shrimp, *Penaeus monodon*), oysters (*Ostrea edulis*) and Atlantic salmon (*Salmo salar*) cultured in marine enclosures, have generated concern over disease and other impacts that might arise from their escape.

3.3 Biodiversity and people

3.3.1 Cultural diversity: How many indigenous groups and languages are represented in the Americas?

Cultural diversity is defined as the spiritual, material, intellectual, and emotional processes and dynamics developed by a social group. It is composed of livelihoods, values, traditions, knowledge, and beliefs centered on nature (Berkes, 2008; Posey, 1999; UNESCO, 2002). Traditional cultural and spiritual values provide the context in which environmental stewardship can be nurtured (Kothari, 2009; Robson & Berkes, 2012).

In the 1980 census, half of the Latin American countries quantified their indigenous populations based on linguistic criteria (CEPAL, 2014; Correa, 2011). As of 2000, 16 out of 19 countries identified their indigenous populations on the basis of self-determination, common origin, territories, and linguistic and cultural dimensions (Bartolomé, 2006; CEPAL, 2014; F. Correa, 2011; International Labour Organization, 1989). Based on these criteria, in 2014, 826 native populations were legally recognized in the Americas (305 in Brazil, 102 in Colombia, 85 in Peru, 78 in Mexico, 39 in Bolivia) and 15 First Nations populations were recognized in Canada and the USA (United Nations Development Programme, 2014). In Latin America in 2010, indigenous peoples numbered about 45 million. Mexico is home to 17 million (15.1% of the total population); Peru, 7 million (24%); Bolivia 6.2 million (62.2%) and Guatemala, 5.8 million (41%) (CEPAL, 2014). In Canada, First Nations population was less than 1 million (2.6% of the total population) and in the USA, 5.1 million (1.7%) (United Nations Development Programme, 2014). In the 2011 census, aboriginal peoples in Canada totaled 1.4 million, or 4.3% of the population. Some 600 First Nations governments or bands with distinctive cultures, languages, art, and music were recognized. In the USA, 566 distinct Native American tribes are recognized by the government as of 2016, including indigenous peoples of Alaska and Hawaii (Federal Register, 2016). In 2010, the US Census Bureau estimated that about 0.8 or 0.9% of the USA population was of native American descent; one-third of that population lives in California, Oklahoma and Arizona (USA quickfacts census, 2012).

Languages underpin ethnobotanical, ethnozoological and ethnoecological knowledge and guides a people´s spirituality and worldview. Indigenous and local knowledge (ILK) is transmitted by language and thus conserving languages is crucial for understanding biodiversity as it relates to human well-being. Over 1,000 indigenous languages are spoken across the Americas. Most of the indigenous American languages in North America are in trouble, dying or already extinct. Other subregions also face language extinction but are somewhat more stable (Chapter 2, Table 2.2).

3.3.2 Cultural and biological diversity: Traditional knowledge and worldviews among the indigenous communities of the Americas

Traditional knowledge. “Traditional knowledge is the ancestral wisdom and the collective and integrated knowledge that indigenous, Afro-descendants, First Nation peoples, and local communities share based in their praxis in the interrelationship human-nature, transmitted from generation to generation” (De la Cruz et al., 2005). Biodiversity has significance to indigenous communities for human nature, culture and spirituality. Traditional knowledge is collective, intergenerational and linked to the right of free determination and worldview (De la Cruz, 2011; Robson & Berkes, 2012). These interrelationships constitute the biocultural heritage of indigenous people that is intimately related to their connection to land and sacred or spiritual places, and influence how people interact with and manage land. A good example is seen in the indigenous Menominee people who inhabit the Great Lakes region (Box 3.3).

Worldview. The "worldview" is the structured group of diverse ideological systems by which a social group understands the universe and the order of systems, knowledge, and interrelationships with nature. (López-Austin, 1990). Recognition of worldview signifies appreciation for a system that has the potential to be less damaging to the environment than many current dominant practices. The worldview is interrelated with territory, nature, religion, politics and the economy (Zolla & Zolla, 2004). Most indigenous populations share principles that derive from their worldview, including the principle of reciprocity, the principle of correspondence between the micro-cosmos and the macro-cosmos and the principle of complementarity, in which the cosmos functions with all of its parts (Zolla & Zolla, 2004).

Box 3.3. The Menominee Nation: an example of indigenous knowledge and practice.

The Menominee Nation is a nation of indigenous people of North America that has existed for thousands of years. Currently situated in Wisconsin (USA), it stewards one of the significant regions of contiguous vestiges of old growth hardwood forest that remain in the Great Lakes Region. The present-day Menominee reservation is only a fraction of the estimated 4.05 million hectares of ancestral lands accessed by the Omaeqnomenewak prior to European contact. Treaties with the USA government between 1817 and 1856 resulted in a large loss of land, down now to approximately 95,313 ha (Omaeqnomenew Masenahekan, 2004). Much of the Menominee forest is old growth due to efforts by early leaders to manage the resource sustainably in a time when land barons were harvesting what they perceived were unlimited supplies of timber. Some 68% of the region was covered by old-growth forests in the late 1800s (Frelich, 1995), but only about 1% of Wisconsin's old-growth forests remain today as a consequence of producing more than 8.26 million cubic meters of timber annually in the late 1800s. Guided by tribal leaders' philosophy for managing forests and processing of forest products, Menominee forested land provides economic benefits not only through sustainable timber harvesting and wood product manufacturing but also through access to culturally important plant and animal species and ecosystems. As a result, the Menominee forest is home to ecosystems not seen in Wisconsin since before the great forest clear-cuts of the 1800s. The current sustainable forest management is a reflection of the worldview of early tribal leaders expressed in the following management goal: *Maintain the diversity of native species and habitats, continue to improve environmental and cultural protection, improve planning efforts, further develop economic opportunities, promote communication, and increase environmental education for the Menominee people, while maximizing the quantity and quality of forest products grown under sustained yield principles* (Menominee Tribal Enterprises, 2012).

For the Otomi people, an indigenous group in Mesoamerica, worldview explains the universe; the origin and destiny of humanity; the origin of their territory and mountains as the source of fertility and force; the dialogues between humans and animals to seal protection; the creation of plants, health, and sickness as a unity among body, soul and land; and the circle of time and space (Galinier, 1997; Pérez, 2008). Humans are integrated with land, animals, plants, and mountains. Well-being consists of finding equilibrium among these parts. "To be fine is to dominate our soul (*ro mui*)" (Pérez, 2008). Among the Kichwa people in Ecuador, the Sumak Kawsay ("good living") is based on a communitarian space, continuous dialogue with Mother Nature or Mother Earth (pachamama), the conservation of ecosystems, different ways to produce knowledge by all members, social organization based on the principle of reciprocity and solidarity (minka, ranti-ranti, makikuna, uyanza). For Manuel Castro (ECUARUNARI, Ecuador), Sumak Kawsay implies social equity, justice, and peace (Houtart, 2014). For Eugenia Choque, *suma jakaña* means to achieve food sovereignty, and for Xabier Albó it denotes to "live together well" (Houtart, 2014). This worldview constitutes an alternative for development and a "cosmic ethic" (Gudynas, 2009, 2011; Houtart, 2014). Much can be learned from the worldview of indigenous peoples when it comes to sustainability and biodiversity conservation.

3.3.3 Domestication and use of biodiversity and agroforestry

Domestication. The northeastern USA, Mesoamerica, the Andean region of Peru, Ecuador and Bolivia, and the Amazon basin are widely recognized as primary sites of management and domestication of biological diversity in the Americas (Casas et al., 2007; Chacón et al., 2005; Clement et al., 2010; Galluzzi et al., 2010; Harlan, 1971; Kwak et al., 2009; Parra & Casas, 2016; Perry et al., 2007; Smith, 1994) (see also Chapter 2).

Many plants were domesticated in Mesoamerica (mainly 30 food species, such as maize, beans, tomatoes, cacao, squash, and chili), and the Andean region (potato, quinoa, squash, maize, beans, chili), Brazil, Paraguay (mate, pineapple, some nuts) (Harlan, 1961; Kloppenburg, 1991; Nemogá Soto, 2011) (see also Chapter 2). In the northeastern USA, native peoples domesticated perhaps 20 plant species, dogs, and turkeys; in the Mesoamerica subregion nearly 200 plant species, dogs, turkeys, and cochineal were domesticated (Casas et al., 2017; Zarazúa, 2016). In the Andean region of Peru, 182 plant species, dogs, and two species of camelids (llamas and alpacas) were domesticated (Wheeler, 2017), as well as the guinea pig and possibly the duck *Cairina moschata* (Torres-Guevara et al., 2017). In the Amazon, at least 80 species of edible plants have been domesticated (Clement, 2017; Clement et al., 2016). In Mexico, incipient management may include 800 to 1,200 plant species, whereas in Peru nearly 1,800 species are incipiently managed (Casas et al., 2016; Casas et al., 2017; De Jong, 1996; Fraser et al., 2011; Moreno-Calles et al., 2016; Moreno-Calles et al., 2016; Peri et al., 2016; Somarriba et al., 2012; Torres-Guevara et al., 2017).

In addition to agricultural development, local populations manage a high diversity of forests (tropical, dry, temperate, boreal) and ecosystems (coastal, wetland, mountain, plain, desert, aquatic) from which they obtain food, medicine, wood, fuelwood, water, tools, handicrafts, colorants, fodder, ornamental, biological control and instruments. Traditional agricultural systems in the Americas, a result of millennia of cultural and biological evolution, harbor high levels of biodiversity, planned and associated, and represent a high-quality matrix that allows forest species movements among patches (Galluzzi et al., 2010; Larios et al., 2013; Perfecto & Vandermeer, 2008). Traditional farming systems can have a structural complexity and multifunctionality that benefit people and ecosystems and allow farmers to maximize harvest security and reap the benefits of the multiple use of landscapes with low-environmental impacts (Altieri, 2000; Galluzzi et al., 2010). For example, Mayan milpa systems, characterized by open field gaps, reforested plots, and mature closed-canopy forests are recognized for their high agrobiodiversity. In Mayan milpa systems of Greater Petén on the Yucatán Peninsula, around 99 cultigens of native species have been reported as dominant plants on the open multi-crop maize fields, and more than 30 native tree species are managed or protected inside the long-lived perennial reforestation plots and under closed canopies (Ford & Nigh, 2015). Saving such biodiversity should be a priority.

Use of biodiversity. Besides domestication, the biologically-diverse Americas contain a large amount of other biodiversity used by people, including plants, vertebrates, arthropods, fungi, lichens, bacteria, and yeasts. For Mexico, the ethnobotanical data bank at the Universidad Nacional Autónoma de México records close to 7,000 useful plant species out of a total of 24,000 for the country (Casas et al., 2017; Casas et al., 2016). Studies in some regions of Mexico indicate that, on average, nearly 40% of plant species are useful. Such information leads to an estimate of around 10,000 useful plants in Mexico. In Peru, different studies have recorded some 4,400 useful plant species (Torres-Guevara et al., 2017). Mesoamerican peoples are known to use about 7,000 plant species, mainly for medicines; 3,000 animal species (including insects); and 120 fungal species (Caballero & Cortés, 2001; Hernández, 1985; Rojas, 1991).

Agroforestry. In Latin America, an estimated 200 to 357 million ha are under agroforestry (Somarriba et al., 2012). About 12 recognizable types are found, seven in the tropics and five in temperate zones (AFTA, 2017; Jose et al., 2012; Kort et al., 2014; Nair, 1985; Nair et al., 2008; Peri et al., 2016; Somarriba et al., 2012). Agroforestry systems in North America and part of southern South America are of recent origin, while central and northern South American agroforestry systems are bound to highly diverse cultural zones, where societies have preserved their traditional knowledge over thousands years (Casas, Parra et al., 2016; Casas, Parra-Rodinel, Rangel-Landa et al.,2017; De Jong, 1996; Fraser et al., 2011; Moreno-Calles, Casas, Rivero-Romero et al., 2016; Moreno-Calles, Casas, Toledo et al., 2016; Somarriba et al., 2012; Torres-Guevara et al., 2017). Ethnoagroforestry management conserves native wild plants, wild and domesticated animals, and the interactions among them (Moreno-Calles et al., 2016; Pell, 1999). Species richness of non-volant mammals and amphibians is similar for agroforestry systems and forests (Chaudhary et al., 2016; Danielsen et al., 2009; García-Morales et al., 2013; Mendenhall et al., 2014; Philpott et al., 2008). However, forest birds, particularly specialist species, and phytophagous bats have declined over time in richness and abundance, respectively, in agroecosystems (Danielsen et al., 2009; García-Morales et al., 2013; Mendenhall et al., 2014; Philpott et al., 2008; Gonçalves et al., 2017).

Agroforestry systems are being lost due to human migration, access to commercial markets, land use change, and the disinterest of government agencies (Montes-Leyva et al., 2017; Van Vliet et al., 2012). The creation of agroforestry systems based on traditional indigenous and local knowledge and novel technological advances promises improvement of ecological interactions, provision of multiple products and ecosystem services (Jose et al., 2012; Moreno-Calles et al., 2016; Moreno-Calles et al., 2016; Peri et al., 2016), and if stimulated, would contribute to biodiversity conservation.

3.3.4 Status and trends of biodiversity in urban anthropogenic systems

Status. Urban areas are home to about 80% of the population in the Americas. Urban land in the North American (excluding Greenland) and Mesoamerican subregions accounts for 5% of the total land (Güneralp & Seto, 2013). The Caribbean subregion has the highest urban land fraction (16%) and South America the lowest (2%). Currently, the Americas host eight (20%) of the world´s 40 Megacities (population over 10 million): two in the North American subregion, one in the Mesoamerican subregion and five in the South American subregion. There are many other large cities in the Americas that do not qualify as megacities (Figure 3.12A). Urban ecosystems in the Americas are expected to continue to expand and coalesce (Seto et al., 2012). This signifies that urban areas will be the main contact point with nature for an increasingly large proportion of the Americas population. Policies that conserve and enhance urban biodiversity will thus enhance human well-being.

Urban areas in many parts of the Americas are surrounded by high-diversity ecosystems. Major changes in species richness, species composition, and ecosystem functioning have accompanied urbanization (McPhearson et al., 2013; Pauchard & Barbosa, 2013) although cities may be hotspots of plant biodiversity because of human cultivation (Müller et al., 2013). A survey of spontaneous and cultivated flora across seven USA cities found a positive association between species richness and urbanization (Pearse et al., 2018), a pattern that has been observed in other regions (Hope et al., 2003; Walker et al., 2009). However, urbanization can lead to loss of spontaneous species richness and phylogenetic diversity and selects for plants with functional traits that allow them to disperse and reproduce well in the urban environment (Knapp et al., 2012). That is, the urban flora is a non-random sample of plant biodiversity.

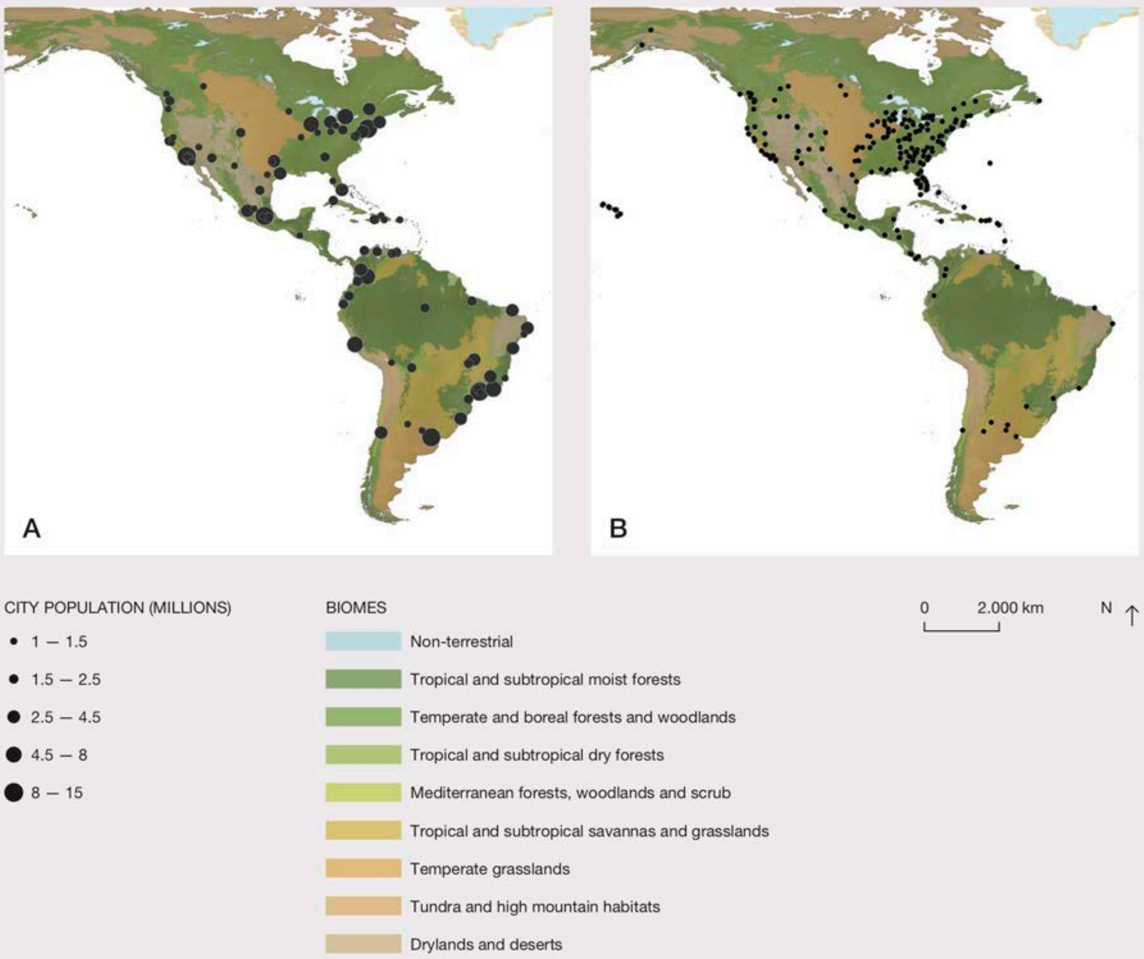
Cultivated plant species in North America, and perhaps across the Americas, include a high number of introduced species (Pearse et al., 2018). Such introduced species can escape cultivation (Knapp et al., 2012; Pearse et al., 2018) and interact with native species, changing the floral composition in urban areas and beyond (Shochat et al., 2010). Indeed, the proportion of exotic plants is expanding, and the number of native species is declining in urban areas in the Americas (Reichard & White, 2001; McKinney, 2002; Kowarik, 2008; MacGregor-Fors & Ortega-Álvarez, 2013), while urban floras are tending to homogenize (La Sorte & McKinney, 2007). Consequently, urbanization affects community assembly and leads to more simplified (Aronson et al., 2014; McKinney, 2002; Stranko et al., 2010) and more homogenized ecosystems (Groffman et al., 2014; Hall et al., 2016; La Sorte & McKinney, 2007; McKinney, 2006; Steele et al., 2014).

Box 3.4. Botanical gardens in the Americas.

otanical gardens are stores of plant biodiversity that provide ex-situ conservation and biodiversity education to urban populations. However, there is a large imbalance in the distribution of botanical gardens across the subregions. Of the 2,728 botanical gardens registered globally with the botanic gardens Conservation International, 765 occur in North America, 127 in Mesoamerica, 46 in the Caribbean, but only 164 in South America (Figure 3.12. B). South America's relatively low number is noteworthy given that it houses higher plant species richness and more megacities than North America. Some very rich biomes, like the South American Mediterranean forests, shrublands and scrub biome, have a very poor representation of certified botanical gardens (Figure 3.12. B).

Figure 3 12 A Largest cities in the Americas based on population size shown by biome. B Location of accredited botanical gardens from the BGCI garden search database in relation to biomes.

Source: A: http://simplemaps.com/data/world-cities, last updated in 2015. B: Search database www.bgci.org. Accessed August 5, 2017.



Some plant and animal species tend to do well in the physical structure of the urban landscape and are able to take advantage of the availability of resources such as human garbage. However, animal species richness tends to decline along urbanization gradients (Aronson et al., 2014; Chace & Walsh, 2006; González-Urrutia, 2009; Groffman et al., 2003; Hamer & McDonnell, 2008; McKinney, 2002, 2008; Moore & Palmer, 2005; Ortega-Álvarez & MacGregor-Fors, 2011; Paul & Meyer, 2001; Stranko et al., 2010; Urban et al., 2006). That said, nonlinear relationships have also been reported for animal species along these gradients (Blair &

Launer, 1997; Faggi & Perepelizin, 2006; Germaine & Wakeling, 2001; McIntyre et al., 2001; McKinney, 2008).

Urban environments are associated with a decline in native mammals, with the rare exception of species able to thrive near humans. Carnivorous and large mammals have been progressively excluded from urban areas, while middle-size omnivorous mammals that eat anthropogenic foods tend to persist (McCleery, 2010; Pereira-Garbero et al., 2013). Many small mammals in the Americas are poorly represented in cities except rats and mice (Cavia et al., 2009; Childs & Seegar, 1986; Himsworth et al., 2013). The response of reptile biodiversity to urbanization is poorly understood, although positive trends were reported for turtles and snakes (Barrett & Guyer, 2008). In Arizona, lizard diversity and abundance follows a humped pattern on a residential density gradient (Germaine & Wakeling, 2001).

Birds are among the most studied urban animals. Avian diversity and urbanization are negatively correlated, while the total abundance of birds may increase with urbanization (Chace & Walsh, 2006; González-Urrutia, 2009; Ortega-Álvarez & MacGregor-Fors, 2011). As in other taxa, these trends are associated with shifts in functional traits along urbanization gradients (Chace & Walsh, 2006; Leveau, 2013; McKinney, 2002) and species ability to use waste as food (Marateo et al., 2013). Urban bird diversity is enhanced by increases in the number, size, connectivity and habitat heterogeneity of urban parks and vegetation remnants (Beninde et al., 2015; Díaz & Armesto, 2003; Garitano-Zavala & Gismondi, 2003; González-Urrutia, 2009; Juri & Chani, 2009; Manhães & Loures-Ribeiro, 2005; Maragliano et al., 2009; Ortega-Álvarez & MacGregor-Fors, 2011; Perepelizin & Faggi, 2009; Sacco et al., 2013; Villegas & Garitano-Zavala, 2010). Significant raptor diversity has been reported, even in larger cities. For example, more than 20 raptor species were recorded in Buenos Aires, Argentina (Cavicchia & García, 2012). Some 24 species (83% of Chilean raptor species) were observed in the Chilean Metropolitan Region of which 18 occur in the vicinity of Santiago; seven are considered urban or suburban (Jaksic et al., 2001). In Baja California, Mexico, raptor richness was unaffected by the anthropogenic transformation of the habitat (Rodríguez-Estrella et al., 1998). At the same time, non-native avian species have progressively established in urban areas. In Mesoamerica some urban areas now have non-native avian abundances similar to those observed in developed countries at temperate latitudes (González Oreja et al., 2007). In the midwestern USA, raptors such as the peregrine falcon, whose populations plummeted with pesticide use in the mid-20th century, have been successfully reintroduced in cities where tall buildings provide suitable nesting sites (Tordoff & Redig, 2001).

Arthropods show a range of responses to urbanization (McIntyre, 2000; Müller et al., 2013, Raupp et al., 2010). In the Phoenix area, for example, birds were found to be a dominant force controlling arthropod ecology (Faeth et al., 2005). While some urban gradients involve small changes in richness or abundance of arthropods, community composition may change considerably (McIntyre et al., 2001). In Palo Alto, California, butterfly diversity has progressively declined with increasing urbanization (Blair & Launer, 1997). However, several studies show a positive relationship between urbanization and some bee guilds (e.g. cavity-nesters within urban areas, Potts et al., 2010).

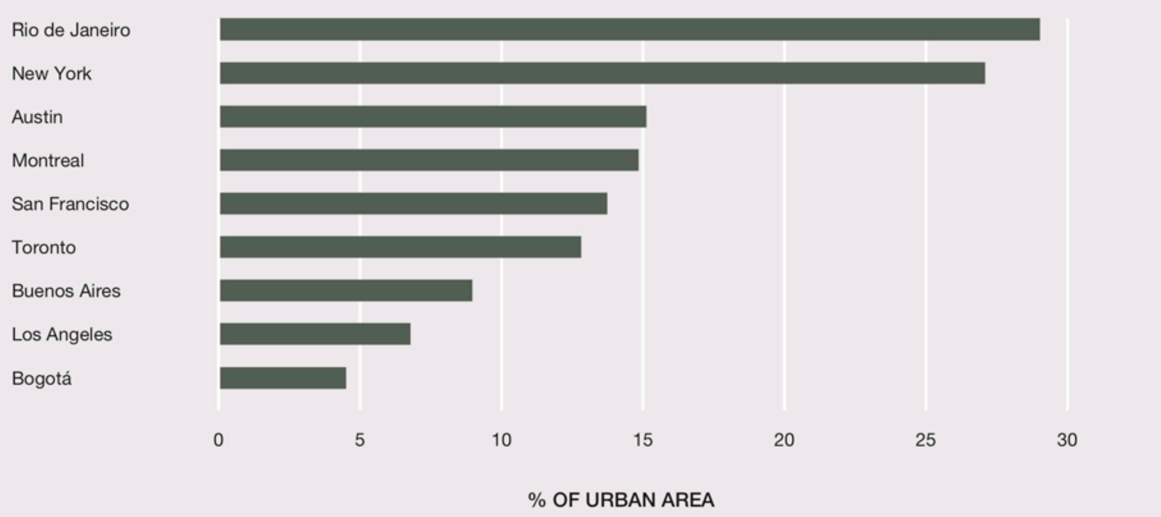
Recent trends. An increase in high-rise buildings has greatly increased population density in many cities of the Americas. Urban ecosystems within these cities have increased in size as the human population has grown (Grimm et al., 2008). This portends large-scale transformations for the provision of water, food, and services (Vörösmarty et al., 2000). Associated transportation systems have created a network of interconnected urban habitats that has grown significantly in extent, density and flow (Kohon, 2011; Rodrigue et al., 2017).

Over the past two decades, the uneven accessibility of urban greenspace has become recognized as an environmental justice issue as awareness of its importance to public health has become recognized (Dai, 2011). Some cities in Latin America have begun to set goals to plan for a minimum of 9m^2 of green area per inhabitant[10]. Data on green areas for cities in the Americas is scarce and this is an area that needs better attention. The percentage of urban areas dedicated to green areas is highly variable across the Americas (Figure 3.13). Considerable variation, moreover, can occur within individual cities. For example one of the

[10] http://ipco.gob.mx/images/documentos/estudios/piam_colima_final_2010.pdf

wealthiest suburbs of Santiago, Chile has 56m^2 per inhabitant, while one of the poorest has only 2.4m^2 (Reyes & Figueroa, 2010). Generally, the incorporation of green areas of any kind has promoted urban biodiversity (Cameron et al., 2012), although the development of green areas has not been commensurate with the population increase in urban areas. Thus, conserving biodiversity in urban areas should be a priority. The establishment of green areas using native species can simultaneously contribute to biodiversity conservation and human well-being and should be a priority.

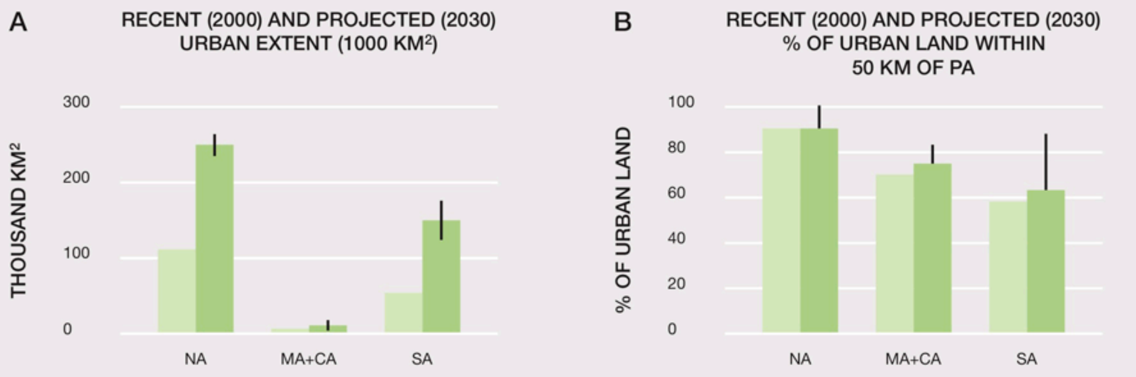
Figure 3 13 **Percentage of urban areas dedicated to green areas in different cities of the Americas. Based on data from World Cities Culture Forum (http://www.worldcitiescultureforum.com/data/of-public-green-space-parks-and-gardens).**



The Americas are projected to experience significant increases in urban land extent (Figure 3.14. a) (Güneralp & Seto, 2013). Moreover, North America is expected to have more than 50% of its total urban lands within 25 km of protected areas and 90% of its urban lands within 50 km of protected areas by 2030; in contrast, South America is projected to have about 65% within 50 km while Mesoamerica and the Caribbean are projected to have somewhat more (Figure 3.14. b) (Güneralp & Seto, 2013). Documented changes in hydrology with urbanization, including alteration of wetlands (Steele et al., 2014), pollution, simplification of freshwater environments and loss of riparian vegetation, will tend to reduce biodiversity among algae, plants, invertebrates and vertebrate communities (Groffman et al., 2003; Moore & Palmer, 2005; Paul & Meyer, 2001; Stranko et al., 2010; Urban et al., 2006). Amphibians are particularly vulnerable to urban development (Hamer & McDonnell, 2008), habitat loss, homogenization and isolation (Birx-Raybuck et al., 2010; Cushman, 2006; da Silva et al., 2011, 2012; Delis et al., 1996; Fahrig, 2003; Fahrig et al., 1995; Sutherland et al., 2010) and changes in hydrodynamics (Barrett et al., 2010; Eskew et al., 2012; Price et al., 2011).

Figure 3 14 A Total urban extent in 2000 (light green) and projected in 2030 (dark green); B percentage of total urban land in 2000 (light green) and projected (dark green) in 2030 within 50 km of protected areas (PA) for North America (NA), Mesoamerica, the Caribbean (MA+CA) and South America (SA).

Source: Modified from figures in main text and supplementary material in Güneralp & Seto (2013).



Long-term data on biodiversity in cities of the Americas still tends to be limited and fragmented. In the USA, two urban Long-Term Ecological Research sites (Baltimore and Phoenix) have been established to gather social and ecological data (Redman et al., 2004). Such Long-Term Ecological Research sites are valuable for the purpose of comparative international research on urban socio-ecological systems and their links to decision-making. The City Biodiversity Index ("Singapore Index"), which integrates biodiversity data, has been widely used in South East Asia to assess the role of cities in conserving biodiversity. This, or some similar index, could be adapted for use in the Americas.

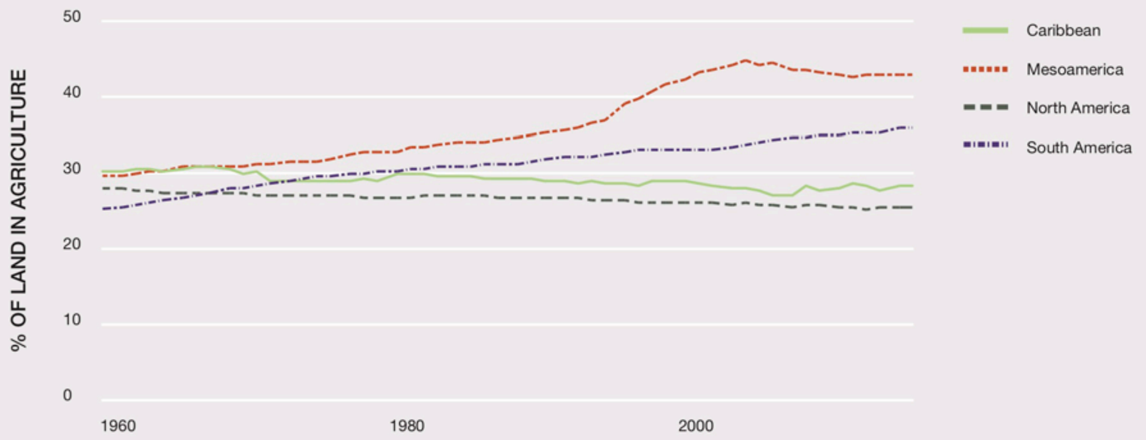
3.3.5 Status and trends of biodiversity in agricultural, silvicultural and aquacultural anthropogenic systems

European colonization simplified agricultural systems and landscapes across the Americas, reducing crop diversity, marginalizing several native crops and eroding knowledge associated with traditional farming practices (Galluzzi et al., 2010; Galluzzi & López Noriega, 2014; Khoury et al., 2014; Kremen & Miles, 2012; Winograd et al., 1999). As a consequence, large amounts of land in the Americas are today devoted to intensive cropping and forestry (c.f., Beddow et al., 2010). Conversion of land from natural systems to crop production and agriculture has important impacts on habitat for biodiversity and differs by biome (Ramankutty et al., 2010) and type of farming system. For Latin America, expansion of pastures is the main cause of habitat loss and is responsible for more than two-thirds of deforestation in the Amazon region, with agrofuel and fodder (soybean) monocultures also adding pressure to forests (Altieri, 2009; Pacheco et al., 2011; Thornton, 2010). Agricultural intensification changes and diminishes ecological functions (Goijman et al., 2015) and can lead regionally to shifts in species composition (section 3.4 for details of impacts in different terrestrial biomes). Traditional knowledge and systems for the maintenance of crop genotypes have been lost as agriculture has been commercialized. For example, there is evidence of a loss of large numbers of native potato in Cusco (Gutiérrez & Schafleitner, 2007), due to the introduction of commercial strains. This is a vast area of knowledge that was not possible to cover in the present assessment and warrants an assessment on its own merits.

Non-native species are often the base of production systems and can impact ecosystem services needed to support production in the long term. Fishes in aquaculture represent a good example, as nearly all countries culture tilapias, carp and trout, none of which are native to the Americas. Although Brazil contains 20% of the world's fish species, aquaculture is based solely on non-native species – some are native to the country but produced beyond their native ranges (I3N, 2016). The same trend is present in silviculture. Pines (*Pinus* spp.) are widely invasive in the southern hemisphere, with at least 16 species that have spread from planting sites into natural or seminatural vegetation (Richardson et al., 1994), while acacias (*Acacia* spp.) and gums (*Eucalyptus* spp.) are either not planted as much or are less aggressive. These taxa, either in plantations or invasions, have been documented as intensive water users; areas invaded with these trees tend to have low economic value and low productivity (Versveld et al., 1998).

Figure 3 15 **Changes in the percentage of land in agriculture for each subregion from 1961 to 2014. Greenland is not included in the calculation for North America.**

Percentages of individual countries were multiplied by their area, summed, and divided by the total area of the subregion. Source: World Bank (2017). World Development Indicators (https://data.worldbank.org/indicator/AG.LND.AGRI.K2) Last updated Date: March 23, 2017.



Recent trends. The Americas have led world production of high-demand agricultural products like soybeans, sugarcane, and cattle meat over the past five decades. During this period, the net agricultural production of the region has grown together with its population (Ramankutty et al., 2002). This has led to increases in the conversion of land to agriculture (Figure 3.15). The apportionment of land to agriculture (aggregated within each subregion) shows the greatest increases in Mesoamerica followed by South America, but recent declines in the Caribbean and North America. The total extent of arable and pasture lands in Latin America has increased at an annual rate (1990–2008) of 0.87% for South America (16.4 million ha) and 0.15% for Mesoamerica (828,000 ha). Pasture land grew by 11.3 million ha (0.14% per year) in South America, while in Mesoamerica it declined 2.7 million ha (−0.17% per year) (Pacheco et al., 2011). Conversion of land for agricultural purposes has often come at the expense of forest, woodland, and other vegetation types (section 3.4).

From 1992 to 2010, richness and phylogenetic diversity of crop production and exports from all subregions have been relatively constant. However, South America and Mesoamerica have higher phylogenetic diversity in crop production than does North America, and Mesoamerica has higher crop species richness than both North and South America (Nelson et al., 2016). In contrast, North America has a higher consumption of species richness than other subregions, even while all subregions have similar phylogenetic diversity in crop consumption (Nelson et al., 2016).

Pollinator-friendly agricultural systems can help maximize crop yields by preserving the pollination services offered by wild bees (Garibaldi et al., 2014; Shaver et al., 2015). Pollinator loss has been particularly rapid in tropical regions (Ricketts et al., 2008) as well as in extensive temperate regions that have experienced drastic land use transformations, like the Pampas of South America (Medan et al., 2011) and the USA Midwest and Great Plains (Koh et al., 2016). The high use of pesticides across the Americas (Liu et al., 2015) is an important additive and interactive cause of bee declines (Goulson et al., 2015).

Aquaculture has increased in the Americas. In the USA, aquaculture growth for marine fish and shellfish has been below the world average, rising annually by 4% in volume and 1% in value (Naylor, 2006). The main marine species are Atlantic salmon, shrimp, oysters, and hard clams, which together account for about one-quarter of total USA aquaculture production. In South America, Chile is now the second largest producer of salmon globally after Norway (Buschmann et al., 2006). Excessive use of antibiotics in Chilean salmon farms have resulted in antibiotic resistance (Burridge et al., 2010), and this trend may be widespread.

3.3.6 Emerging diseases and biodiversity

Emerging infectious diseases have become a major concern (Hatcher et al., 2012). Bacteria, viruses, protozoan, fungi, helminths and drug-resistant microbes are commonly reported in emerging infectious diseases outbreaks worldwide affecting a wide taxonomic spectrum (Jones et al., 2008; Pedersen et al., 2007). Multiple mechanisms and causes for emerging infectious diseases have been recognized, including biodiversity loss, land use change, urbanization, climate change, human demographics, international travel and commerce, species invasions, pollution, microbial adaptation, war and famine, poverty, and breakdown of public and animal health measures (Hatcher et al., 2012; Jones et al., 2008; Loh et al., 2015). Individually or synergistically, these causes affect patterns of species distributions and favor invasions of reservoirs, hosts, vectors, and pathogens affecting native species (Keesing et al., 2010; Suzán et al., 2009).

Emerging infectious diseases are reported in marine and terrestrial ecosystems and are responsible for several species and populations extinctions worldwide. Coral reef fragmentation, pollution, and warming have favored toxins and pathogens like *Serratia marcescens* (white pox disease) and Vibrio AK-1 (coral bleaching), producing widespread coral reef mortality (Sutherland et al., 2010; Vega Thurber et al., 2014). Likewise, marine mammals have been threatened by morbilliviruses, poxviruses, and papillomaviruses globally (Harvell, 1999). In terrestrial systems, plant communities have been decimated by emerging infectious diseases such as Dutch elm disease (*Ophiostoma* spp.), chestnut blight (*Cryphonectria parasitica*), and jarrah dieback (*Phytophthora cinnamomi*) that affects hundreds of host plants (Anderson et al., 2004). Several examples of emerging infectious diseases have been reported to affect vertebrates, including *Batrachochytrium dendrobatidis*, a fungal infection producing population and species extinction in amphibians worldwide, and malaria infection in Hawaiian birds (Smith et al., 2009). In the Americas, several endangered and threatened species have declined as a result of emerging infectious diseases such as West Nile virus in native birds (Robinson et al., 2010; Smith et al., 2009), plague in prairie dog colonies (Stapp et al., 2004) and White-nose syndrome in North American bats (Frick et al., 2017). Several infections affect top predators, including canine parvovirus in wild carnivores (Pedersen et al., 2007) and canine distemper, which is associated with extinction in the wild of the black-footed ferret (McCarthy et al., 2007; Thorne & Williams, 1988). Increasing spread of infectious diseases can be expected with globalization, calling for greater vigilance.

3.4 Status and recent trends of biodiversity by units of analysis

3.4.1 Terrestrial biomes

In this section, snapshots of the status and recent trends in biodiversity for the major terrestrial biomes are examined in each subregion where they occur (see Chapter 1 for official units of analysis map of the assessment). Although coverage is extensive, space limitations prevented assessment of all biomes in each subregion and exhaustive treatments for the biomes that are assessed. Status and recent trends in biodiversity and the relative importance of NCP are synthesized in Figures 3.24 and 3.25, respectively. Summary data on species richness for the biomes assessed in each subregion can be found in Table 3.4.

3.4.1.1 Tropical and subtropical moist forests

Mesoamerican subregion

Status. Species diversity in the Mesoamerican broad-leaved tropical/subtropical moist broadleaf biomes is high, with low to moderate species endemism (Myers et al., 2000; Ray et al., 2006). In Mexico, moist wet forests and montane cloud forests have the highest diversity of plant species per unit area among vegetation types (Rzedowski, 1991). Tropical lowland broadleaf moist forests house around 17% of the flora of Mexico, while montane mesophyll forests contain around 9% of the flora (see also, Table 3.4 for numbers) (Challenger & Soberón, 2008). Mesoamerican coniferous forests in general support low to moderate species diversity. Notably, however, Mexican coniferous forests contain very high numbers of pine and oak species (Table 3.4). Species diversity and endemism for amphibians are high in the moist forests of the Mesoamerican highlands (Köhler, 2011; Lamoreux et al., 2015). In Mesoamerican lowland rainforests, the diversity of mammals decreases from eastern Panama to southern Mexico (Voss & Emmons, 1996). The mesic forests of southeastern Mexico have been classified as critically endangered (Hoekstra et al., 2005).

Recent trends. Over the past 50 years, loss of lowland moist forest in Mexico was acute, the yearly deforestation rate reaching 2.6% for 1976-1993 and 1.3% for 1993–2002 (Challenger & Dirzo, 2009); by 2002 primary forest was down to only 17.5% of the original area. Before the late 1980s, forest loss was generally caused by small-scale slash-and-burn agriculture. In the past 25 years, however, large-scale cropping and pastures became the main causes of tropical habitat loss (Gibbs et al., 2010; Laurance, 2010). Montane mesophyll forest (including cloud forest) was reduced from less than 50% to 28% of its original extent over the period 1976 -2003; coniferous forests fared better, with around 50% still remaining (Challenger & Dirzo, 2009).

Removal and fragmentation of moist forest have led to a significant decrease of regional species diversity (Ray et al., 2006). Many amphibian species have experienced severe local and regional declines across the moist forests of the Mesoamerican highlands due to habitat destruction, emerging infectious diseases and other factors (Lamoreux et al., 2015; Stuart et al., 2008). The increased use of pesticides and fertilizers, loss of live fences, and decline of natural habitat fragments within agroecosystems – have also exacerbated biodiversity losses due to habitat reduction (The Nature Conservancy, 2005).

In general, tropical forests seem to be resistant to the impacts of invasive plant species (Denslow & DeWalt, 2008), and compared with habitat loss and fragmentation, exotic invasive species are considered a relatively minor threat to moist forest biodiversity as seen in Mexico (Challenger & Dirzo, 2009; Dirzo & Raven, 2003). Of the 42 exotic species reported by Rejmánek (1996), most are confined to pastures, clearings, or other highly disturbed sites (Foster & Hubbell, 1990; Hammel, 1990). However, there is some evidence that invasive species are increasing (Aguirre-Muñoz & Mendoza, 2009; Espinosa & Vibrans, 2009). The Asian house gecko, *Hemidactylus frenatus*, has been widely introduced in Mesoamerica and is replacing the native leaf-toe gecko, *Phyllodactylus tuberculosus*, especially along the forest edge and in disturbed forests (G. Köhler unpubl. data). It is known to carry the pentastomid parasite, *Raillietiella frenata*, native to Asia, and has been shown to transfer this parasite to *Rhinella marina*, a toad native to Mesoamerica (Kelehear et al., 2015). Several species of Caribbean frogs of the genus *Eleutherodactylus* have been documented as invasive species in Mesoamerican Tropical/Subtropical Moist Broadleaf Forests (Crawford et al., 2011; Köhler, 2008).

Caribbean subregion

Status. The tropical moist forest biome is thought originally to have covered around 81,000 km^2 in the Caribbean (Dinerstein et al., 1995). As of European colonial times and especially before the 1900s (Gould et al., 2012; Lugo et al., 2012), much forest was cleared for agriculture (Fitzpatrick & Keegan, 2007). Dinerstein et al. (1995) estimate that 50% of the original wet forest in the Greater Antilles (90% in Jamaica and Hispaniola) and 25% in the Lesser Antilles was removed or degraded. Land too steep or distant from coastal markets was often left untouched and today forms the core of the remaining biodiversity in Caribbean islands. Vegetation at higher altitudes on the islands of the Lesser Antilles was often retained for "attraction of the rains" (Fitzpatrick & Keegan, 2007; Lugo et al., 2012).

In general, endemism is high for plants and vertebrates in the Caribbean subregion, as is plant species richness. The biodiversity data for the subregion (Table 3.4) to some extent correlates with Caribbean tropical moist forest extent, given that this biome contains a high proportion of Caribbean terrestrial biodiversity. In the Lesser Antilles, the upland moist forests are more species diverse and host the majority of the endemic plant species due to biogeographic factors and human deforestation of the lowlands (Adams, 1997). On the other hand, montane moist forests in the Dominican Republic appear to have lower rates of species richness and endemism than do dry forests (Cano-Ortiz et al., 2015). Cuban invertebrates seem to show high endemism levels similar to those found in vertebrates (e.g. Alayo, 1974; Alayón García, 1999; Starr, personal communication). Among those assessed, some 316 species of plants and vertebrates in the Caribbean are considered threatened (Anadón-Irizarry et al., 2012; IUCN, 2017).

In the Caribbean, terrestrial habitats, including productive areas, are affected by a multitude of invasive alien species — among them, agricultural pests that were introduced with crops. Adverse impacts of invasive species are most severe in the Greater Antilles and Northern Lesser Antilles on islands that have been isolated for the longest time periods and have the greatest degree of human degradation and disturbance (Kairo et al., 2003). However, there is speculation that some invasive exotic plants may act as nurse plants for native species and will decline in importance once native species recover from human disturbance (e.g. *Leucaena* in Puerto Rico) (Lugo et al., 2012). There is also growing acceptance that exotic species have become important components of many island ecosystems (Lugo et al., 2012).

Recent trends. Forests at mid- to high altitudes began to regenerate when agriculture declined after World War I (Gould et al., 2012; Lugo et al., 2012). In Puerto Rico, forest cover increased from approximately 5% to over 30% between 1940 and 1990 (Aide et al., 2000). Tropical moist forest tends to regrow in mountainous areas where agriculture is more likely to be small scale (Asner et al., 2009); 2,550 km^2 of mountain tropical moist forest regenerated between 1984 and 2002 in the Dominican Republic (Grau et al., 2008) and 1,036 km^2 of dry/moist/wet mountain forest regenerated between 1991 and 2000 in Puerto Rico (Parés-Ramos et al., 2008). The area of Caribbean forests in general increased by an average of 0.81% between 1990 and 2010 (FAO, 2011), but in Puerto Rico, the increase has now ceased (Grau et al., 2008). Loss of native predators and herbivores due to introduced predators in post- and pre-Columbian times continues (Kairo et al., 2003).

South American subregion

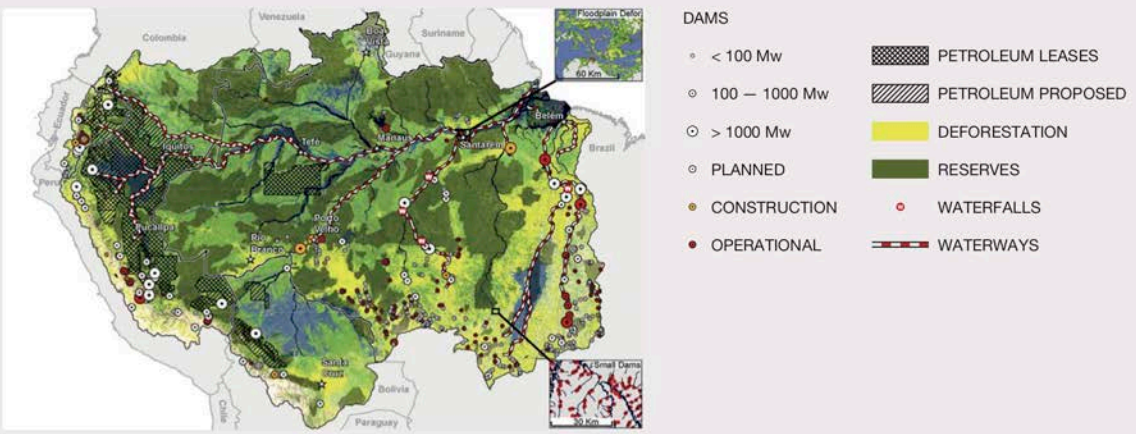
Status. In South America, this biome is centered on the Amazonian wet forest, Atlantic coastal forest, and Andean tropical montane forest. It is also found on the western side of the northern Andes (at low altitudes) and in lowland Venezuela, Guyana, Suriname, and French Guiana. Amazonian wet forest covers 6.7 million km^2 — half of the planet's remaining tropical forests. Around 17% of Amazonian wet forest has been destroyed (Charity et al., 2016) (see also Figure 3.16). Andean tropical montane forest comprises cloud forests (northern Andean forests, Yungas forests and Bolivia-Tucuman forests) and seasonal (wet) forest mostly found above 1,500 m.a.s.l. Atlantic coastal forest once covered around 1.5 million km^2 but today is down to ~12% of its original pre-colonial extent (Ribeiro et al., 2011). Continuous expanses of forest (measured as the proportion of forest more than 1 km from the forest edge) have decreased from 90% (historical) to 75% (today) in Amazonian wet forest and from 90% to less than 9% in Atlantic coastal forest (Haddad et al., 2015). The Amazon, long thought to be a pristine forest, is now recognized as having been subject to long-standing indigenous management and transformation (Roberts et al., 2017). At least 138 crops in 44 plant families, mostly trees or woody species, were cultivated, managed or promoted in Amazonia upon European contact, although some were subsequently lost (Clement, 1999). Human use of biodiversity has been associated with the origin of many new varieties of manioc in both Amazonian wet forest and Atlantic coastal forest (Emperaire & Peroni, 2007).

Exceedingly rich (Table 3.4), Amazonian wet forest is estimated to house one-tenth of all known species of plants and animals (Charity et al., 2016), although these estimates require careful verification. Although opinions differ widely regarding total tree species richness (Table 3.4), it seems that relatively few species account for the bulk of the Amazonian wet forest trees (ter Steege et al., 2013, 2016). Also very rich, Atlantic coastal forest has high endemism (Kier et al., 2009; Mittermeier et al., 2005; Tabarelli et al., 2010). For example, 16–60% of birds, mammals, reptiles, and amphibians in Atlantic coastal forest are endemic (Mittermeier et al., 2005; Tabarelli et al., 2010). Andean tropical montane forest likewise has many range-restricted species (Fjeldså & Rahbek, 2006), high bird species diversity (Table 3.4), high species turnover along altitudinal gradients and high endemism. Epiphytes, which have high water storage, are especially abundant in Andean tropical montane forest (Brown, 1990; Kessler, 2001; Kramer et al., 2005; Krömer et al., 2006; Küper et al., 2004; Roque & León, 2006), as they are in Atlantic coastal forest (2,256 species of hemi-epiphytes, the equivalent of 15% of all vascular plants in these forests) (Freitas et al., 2016)**.**

South American tropical and subtropical moist forests provide important biodiversity-linked NCP. Amazonian wet forest stores 10% of global carbon and places seven trillion tons of water per year into the atmosphere, contributing to the stabilization of local and global climate and nurturing agriculture (Charity et al., 2016). Although globally less relevant than Amazonian wet forest, mature Andean tropical montane forest has higher above-ground biomass than was originally thought (Spracklen & Righelato, 2014). Slope stability, critical in Andean countries, is higher in secondary Andean tropical montane forest than in forest land converted to pastures (Guns & Vanacker, 2013). Pollination provided by wild bees and birds, and animal dispersal are additional biodiversity-linked ecosystem services provided by this biome (see Box 3.5). Currently, many orchids in Ecuador are grown commercially (Mites, 2008), and orchid greenhouses are now a major tourist attraction.

Recent trends. Deforestation rates in the Amazon decreased during the past decade but increased again as of 2015 (RAISG, 2015). Habitat loss in Atlantic coastal forest remains high in most regions, attaining annual rates of 0.5% for the whole biome (Teixeira et al., 2009, see also Chapter 4). Between 2000 and 2012, the net loss of Atlantic coastal forest was proportionally lower than for other tropical woody biomes (Figure 3.19), but this is considered to be due mainly to the establishment of exotic tree plantations (Salazar et al., 2015). Andean tropical montane forest was lost in all Andean countries between 2005 and 2010: between 1985 and 2000 Colombia lost close to one million ha of montane forest (Tejedor Garavito et al., 2012).

Figure 3 16 **The Amazon basin showing basin-wide deforestation (including all areas classified as under human use in both forests and savannah ecosystems), main waterways and river channel network, protected areas, hydroelectric dams, areas available to be leased for oil exploration, and proposed areas for future lease for oil exploration. Source: Castillo *et al.* (2013).**



Deforestation has impacted tree species in Andean tropical montane forest, judging by the 235 species classified as globally threatened according to the International Union for Conservation of Nature (IUCN) Red List of Categories and Criteria (Tejedor Garavito et al., 2014). Upon taking recent deforestation into account, some Andean species representing different taxonomic groups in the IUCN lists were judged as requiring updating in terms of extinction risk (Tracewski et al., 2016), suggesting heightened impacts. Reductions in habitat and biodiversity in Andean tropical montane forest are in part due to down-burning fires set in páramo and puna (e.g. Román-Cuesta et al., 2011). Ongoing deforestation is affecting range sizes (Peralvo et al., 2005; Ocampo-Peñuela & Pimm, 2015), genetic connectivity among populations (Klauke et al., 2016) and stream quality (Iñiguez-Armijos et al., 2014). Moreover, hydrologic connections between the atmosphere and surface waters and their downstream effects have been altered in Andean tropical montane forest - soil moisture can be significantly lower in pasture compared with forest (Ataroff & Rada, 2000).

Forest fragmentation has been associated with long-term losses in species richness and changes in species composition (Haddad et al., 2015; Metzger, 2009; Laurance et al., 2017). In Atlantic coastal forest, old-growth forest patches operate both as irreplaceable habitats for forest-obligate species and as stable source areas (Tabarelli et al., 2010). Fragment size distribution, structural connectivity, matrix quality, remaining forest cover, presence of old-growth forest patches and/or proportion of edge-affected habitats have been identified as key correlates of species richness and abundance in bats, reptiles, birds, canopy/emergent trees, small mammals, mammalian carnivores, butterflies, chironomid insects, and frogs (Tabarelli et al., 2010). Multi-taxa data collected at regional and local scales in the northern Amazon demonstrate reduced species richness with increasing anthropogenic disturbance and considerably more biotic homogenization in arable croplands and cattle pastures than in disturbed, regenerating and primary forest (Solar et al., 2015). Likewise, multi-taxa studies reveal a threshold forest cover that triggers local extinctions (Joly et al., 2014). A survey of a wide range of taxa within a large forest mosaic recorded only about 50% of old-growth forest species richness within patches of tree plantations (*Araucaria, Pinus* and *Eucalyptus*) (Fonseca et al., 2009). Overall, habitat degradation has driven a fraction of Atlantic coastal forest´s unique biodiversity to near extinction (Joly et al., 2014; Tabarelli et al., 2010). Nevertheless, landscape dynamics suggest young secondary forests are beginning to expand in the Amazon, reducing forest isolation and maintaining a significant amount of the original biodiversity (Lira et al., 2012). On the other hand, reduction of traditional practices in Atlantic coastal forest has led to the local loss of cultivar varieties (Peroni & Hanazaki., 2002).

Box 3.5. Nature´s contributions to people (NCP) of the South American Atlantic coastal forest.

Reflecting the very high NCP contribution of tropical and subtropical moist forest (Figure 3.25), the importance of the Atlantic coastal forest goes beyond its rich and diverse biota. First, Atlantic coastal forest provides water for 125 million people, representing three-quarters of Brazil's population and for electricity production. Additionally, Atlantic coastal forest provides food. The fruits of the Myrtaceae species, palms, legumes, and passion flowers are important components of the diet of traditional and local people, while other species provide raw materials such as fibers and oils. Many traditional populations rely on Atlantic coastal forest vertebrates as a source of protein. This part of the more inclusive tropical and subtropical moist forest biome plays an important role in climate regulation and soil stability. Disrupting this stability signifies increased landslides and floods, with disastrous consequences for human populations. In terms of agriculture-related NCP, Atlantic coastal forest hosts some 60 species of Euglossini bees, known to be long-distance pollinators. Finally, the cultural value of Atlantic coastal forest dates back >8,000 years. Atlantic coastal forest remnants are increasingly important for recreation in urban areas, where they serve as parks or urban forests (Joly et al., 2014).

Overharvesting in Amazonian wet forest has caused recent declines in animal populations and basinwide collapse in aquatic species (Antunes et al., 2016). Likewise, many species have proven susceptible to road kill, predation or hunting by humans near roads (Laurance et al., 2009). Hunting of large mammals that disperse seeds of many Neotropical trees can lead to important losses in above-ground biomass (Peres et al., 2016). Defaunation thus has the potential to erode carbon storage, even when only a small proportion of large-seeded trees are extirpated (Bello et al., 2015). The conservation of large frugivorous vertebrates is therefore important to reduce emissions from deforestation and forest degradation.

3.4.1.2 Tropical and subtropical dry forests

Mesoamerican subregion

Status. Tropical and subtropical dry forests are rich in biodiversity, particularly insects, as seen for data for mostly northwestern Costa Rica and Mexico (Table 3.4). The flora of Mexican lowland dry forests shows outstanding endemism (25% at the generic level and 40% the species level) (Challenger & Soberón, 2008). An estimated 72% of this biome, found mostly along the Pacific side of the Mesoamerican subregion, from Panama to western Mexico, is converted (Portillo-Quintero & Sánchez-Azofeifa, 2010). Today Tropical and subtropical dry forests are considered among the most threatened of all terrestrial ecosystems worldwide (Calvo-Alvarado et al., 2013; Janzen, 1988; Frankie et al., 2004). Mexico contains the largest remaining extent in the Mesoamerican subregion (181,461 km^2) (Portillo-Quintero & Sánchez-Azofeifa, 2010).

Tropical and subtropical dry forests have attracted far less attention than tropical moist forests. Not surprisingly, comprehensive information on population trends is less abundant. However, several large mammals have gone locally extinct, including the greater anteater (*Myrmecophaga tridactyla*) from Costa Rica (Janzen, 2002). For the dry forests of Mexico, seven mammals, one reptile, and seven birds have been reported as extinct: twelve plant species have been registered as extinct in states of Mexico dominated by dry forest extinct (Baena & Halfflter, 2008; Flores-Villela & Gerez, 1994). For the Chamela-Cuixmala region of Mexico, at least 40 vertebrate species (fishes not included) are at risk of extinction, representing about 15% of the regional vertebrate diversity (Ceballos et al., 1993).

More open Tropical and subtropical dry forests is more susceptible to invasion than closed moist tropical forest. Invasive species, especially plants, abound. In Chamela, Jalisco, Mexico, 20 exotic species from seven families of plants have been recorded, the grass family (Poaceae) being amply represented, along with three exotic animal species, one rodent (*Mus musculus*) and two birds (*Bubulcus ibis* and *Passer domesticus*) (CONABIO, 2016). For Yucatan forests, 90 species of plants from 28 families have been registered as exotic (again, the most species-rich family is Poaceae, followed by legumes) as well as 18 species of animals, including three birds, one rodent and five reptiles (CONABIO, 2016).

Recent trends. Tropical and subtropical dry forests in Mesoamerica have disappeared rapidly over the past 50 years (Bawa et al., 2004; Janzen, 1988). The deforestation rate in Mexico was estimated to be 0.5% per year for the period 1993–2002; by 2002 only 26% of the original cover, by the authors' definition, remained, and only 38% of that is considered to be old-growth forest (Challenger & Dirzo, 2009). Most of this deforestation may be attributed to conversion to pastures and agricultural crops (Masera et al., 1995). However, a major effort to promote natural regeneration of Guanacaste dry forest is ongoing (Calvo-Alvarado et al., 2009) and should serve as a stimulus for other countries in the Mesoamerican subregion for the recuperation of this biome. In the 1970s, the scarlet macaw (*Ara macao*) still occurred in the Guanacaste Conservation Area (Janzen, 2002); reintroduction can be expected in the future as forests regenerate.

The Africanized honeybee (*Apis mellifera*) arrived in the Guanacaste Conservation Area in the early 1980s and now is a low-density member of the local bee fauna (Janzen, 2002). In the 1990s, wild native bee diversity and abundance severely declined throughout Guanacaste Tropical and subtropical dry forests; this is thought to be a possible consequence of reduced flower abundance due to the elimination of pastures and forest not counterbalanced by Tropical and subtropical dry forest restoration (Janzen, 2002). The flammable African pasture grass jaragua (*Hyparrhenia rufa*) has now reached high abundance in Guanacaste, increasing fire frequency with complex impacts on biodiversity (Bonoff & Janzen, 1980; Janzen, 2002; Janzen & Hallwachs, 2016).

Caribbean subregion

Status. Some 92% of the areas suitable for Tropical and subtropical dry forests in the Caribbean are found in Cuba and the Dominican Republic, a total of 124,488km^2, which is close to 9% of this biome in Latin America overall (Portillo-Quintero & Sánchez-Azofeifa, 2010). Around 66% of Tropical and subtropical dry forests has been converted to nonforest in the Caribbean (66% in Cuba, 78% in Haiti, 58% in the Dominican Republic, 54% in Jamaica and 64% in the Cayman Islands) (Portillo-Quintero & Sánchez-Azofeifa, 2010).

In the insular Caribbean, a typical island pattern of moderate to low species richness (Table 3.4) but high species endemism is observed in Tropical and subtropical dry forests (Banda-R et al., 2016). The endemism rate in this biome´s woody plant species is 77.5% in the insular Caribbean (Linares-Palomino et al., 2011). Mirroring the poor conservation state of Caribbean ecosystems, available data show a large proportion of species in Tropical and subtropical dry forests to be vulnerable to extinction or under a greater threat level according to IUCN Red Data List criteria (IUCN, 2017). Terrestrial and freshwater Tropical and subtropical dry forests ecosystems include 51 threatened plant species, 108 threatened reptile species, 16 threatened amphibian species, 35 threatened birds species and four threatened mammal species (IUCN, 2017).

In pre-Columbian times, humans altered habitats using fire and shifting cultivation – especially in Tropical and subtropical dry forests where soils are fertile. Humans also caused the extinction of large mammal species by overhunting or modifying habitat (Fitzpatrick & Keegan, 2007). In European colonial times large areas of this biome were cleared for agriculture in the insular Caribbean, and by the start of the 1900s Tropical and subtropical dry forests on most islands had been largely cleared or degraded (Fitzpatrick & Keegan, 2007; Gould et al., 2012; Lugo et al., 2012).

Recent trends. As mentioned earlier, the Caribbean forest area (both Tropical and subtropical dry forests and moist forests) increased by an average of 0.81% between 1990 and 2010 (FAO, 2011) as agriculture declined on most islands, domestic energy requirements shifted to imported fossil fuels, living standards increased and population levels stabilized or declined and people moved to urban centers from rural areas (Walters & Hansen, 2013). In Puerto Rico, forest cover increased from approximately 5% to over 30% between 1940 and 1990, particularly Tropical and subtropical dry forests (Aide et al., 2000; Ramjohn et al., 2012). However, during the same period urban expansion and tourism lead to declines in Tropical and subtropical dry forests in coastal areas (Gould et al., 2012; Lugo et al., 2012). Notwithstanding, some local declines of the last kind, Caribbean dry forest seems to be on the way to recuperation.

South American subregion

Status. The definition of Tropical and subtropical dry forests in South America lacks consensus (Banda-R et al., 2016; Portillo-Quintero & Sánchez-Azofeifa, 2010; Salazar, et al., 2015). Some authors include the Caatinga and Chaco in tropical and subtropical dry forests while others do not. This makes assessing this biome difficult in South America. The biome scheme adopted by the Americas assessment considers dry Chaco as part of tropical and subtropical savannas and grasslands (3.4.1.6), while Caatinga is considered under drylands (3.4.1.8).

Species diversity in South American Tropical and subtropical dry forests is moderate to high with high species endemism (Table 3.4) (Banda-R et al., 2016; Linares-Palomino et al., 2011; Ojeda et al. 2003; Pizano & García, 2014; Sandoval & Barquez, 2013). According to one source, between 45-95% of Tropical and subtropical dry forests in the Andean countries has now been converted (Venezuela, 74%; Colombia, 67%; Ecuador, 75%; Peru, 95%; Bolivia, 45%) (Portillo-Quintero & Sánchez-Azofeifa, 2010). The figure for Bolivia is likely to include some Chaco. However, another source for Colombia suggests a greater loss at more than 90% (Gómez et al., 2016; Pizano & García, 2014). Some 58 species of amphibians found in Colombian dry forest have been assessed to be at some level of risk; many mammals likewise are at risk (Pizano & García, 2014).

Recent trends. Reflecting the poorer state of knowledge of tropical and subtropical dry forests compared to moist forests (c.f. 3.4.1.1), little data is available on recent trends in this biome in South America. The biome in Eastern Andean Colombia now shows one of the highest fragmentation levels among all vegetation types (Armenteras et al., 2003). Deforestation rates have descended notably of late in Ecuador (Ministerio del Ambiente, 2014; Sierra, 2013). However, over the period 1990-2008 some 31% of the remaining 4985 km^2 of dry and semi-dry coastal forest was removed (Sierra, 2013). For Venezuela, 88% of 3522 km^2 of Maracaibo Tropical and subtropical dry forests was lost between 1985 and 2010 (Morón Zambrano et al., 2015). These data attest to a general tendency for very high deforestation rates in Tropical and subtropical dry forests in South America (Armenteras & Rodríguez Eraso, 2014) and are of great concern given the high NCP contribution of this biome (Figure 3.25).

3.4.1.3 Temperate and boreal forests and woodlands

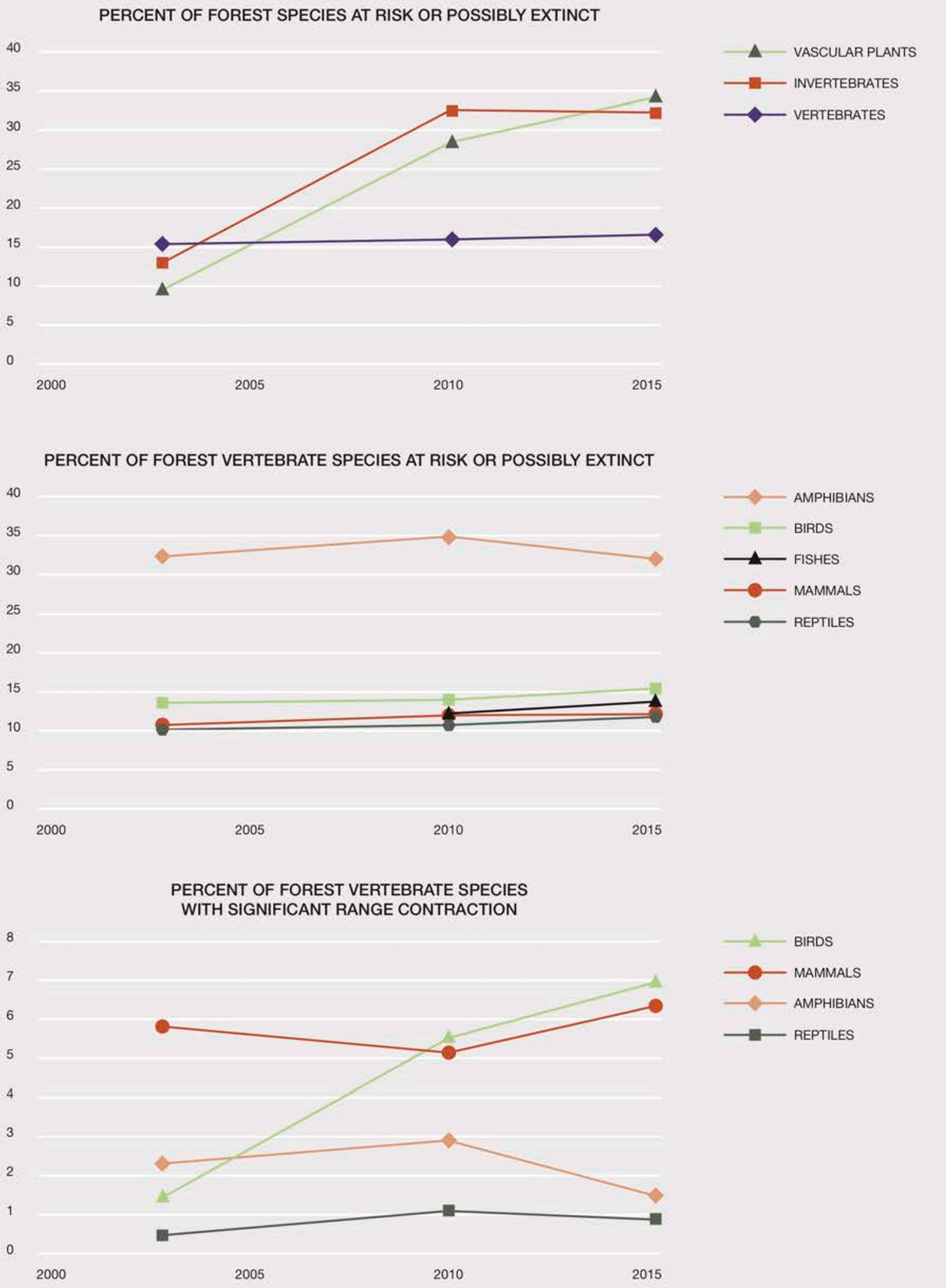
North American subregion

Status. Temperate and boreal forests in North America cover most of the eastern USA and Canada and the Pacific Northwest. Boreal forests, which include many coniferous tree species, occur in colder regions, while deciduous hardwood forests occur in both cold and warm temperate regions. Temperate forests occupy ca. 70% of the land area that was forested at the time of European settlement (Flather et al., 1999). Large numbers of plant and animal species depend on these forest habitats. An estimated 90% of the resident or common migrant vertebrate species in the USA (Flather et al., 1999), and likely in Canada, use forest habitats. The number of forest-associated species is highest in the Southeast and in the arid ecoregions of the Southwest (U.S. Forestry Service, 2015).

Several natural forest types and numerous species have been greatly reduced by human activities. For example, longleaf pine, and loblolly and shortleaf pine forests now cover less than 2% of their presettlement ranges (Noss et al., 1995). Less than 1% of North American temperate deciduous forest has not experienced anthropogenic disturbance (Frelich & Reich, 2009). Temperate deciduous forests have a smaller fraction of original primary forest remaining than do boreal or tropical forests, although most of the original species remain present (Frelich, 1995); 94% of forest-associated vascular plants fully occupy their former range (Nelson et al., 2016). Logging, grazing, fire suppression and manipulation of wildlife populations have altered forest composition, structure, and landscape. An estimated 32% of amphibian species and 12–15% of mammals, birds, reptiles, and fish are possibly extinct or at risk of extinction in USA forests (Nelson et al., 2016). In addition, 32–34% of vascular plants and select invertebrates are possibly extinct or at risk of extinction (Nelson et al., 2016) (Figure 3.17).

Figure 3 17 **Trends in the percentage of forest-associated species determined to be possibly extinct or at risk of extinction.**

Source: Based on Nelson *et al*. (2016), using data from NatureServe (http://www.natureserve.org).



North American forests sequester large amounts of carbon. In the USA, the highest carbon stock densities (> 80 Mg/ha) are found in the upper Lake States, Pacific Northwest, northern New England and coastal areas of the southeastern USA (Heath et al., 2011). Kurz et al. (2013) estimated carbon stock densities above 200 Mg/ha in many managed boreal forests of Canada. However, these values cannot be directly compared because the Canadian estimates included carbon in dead wood and soil. Temperate forests also absorb significant levels of air pollution, including particulate matter, nitrous oxides, sulfur dioxide and ozone, providing benefits to human health (Nowak et al., 2013).

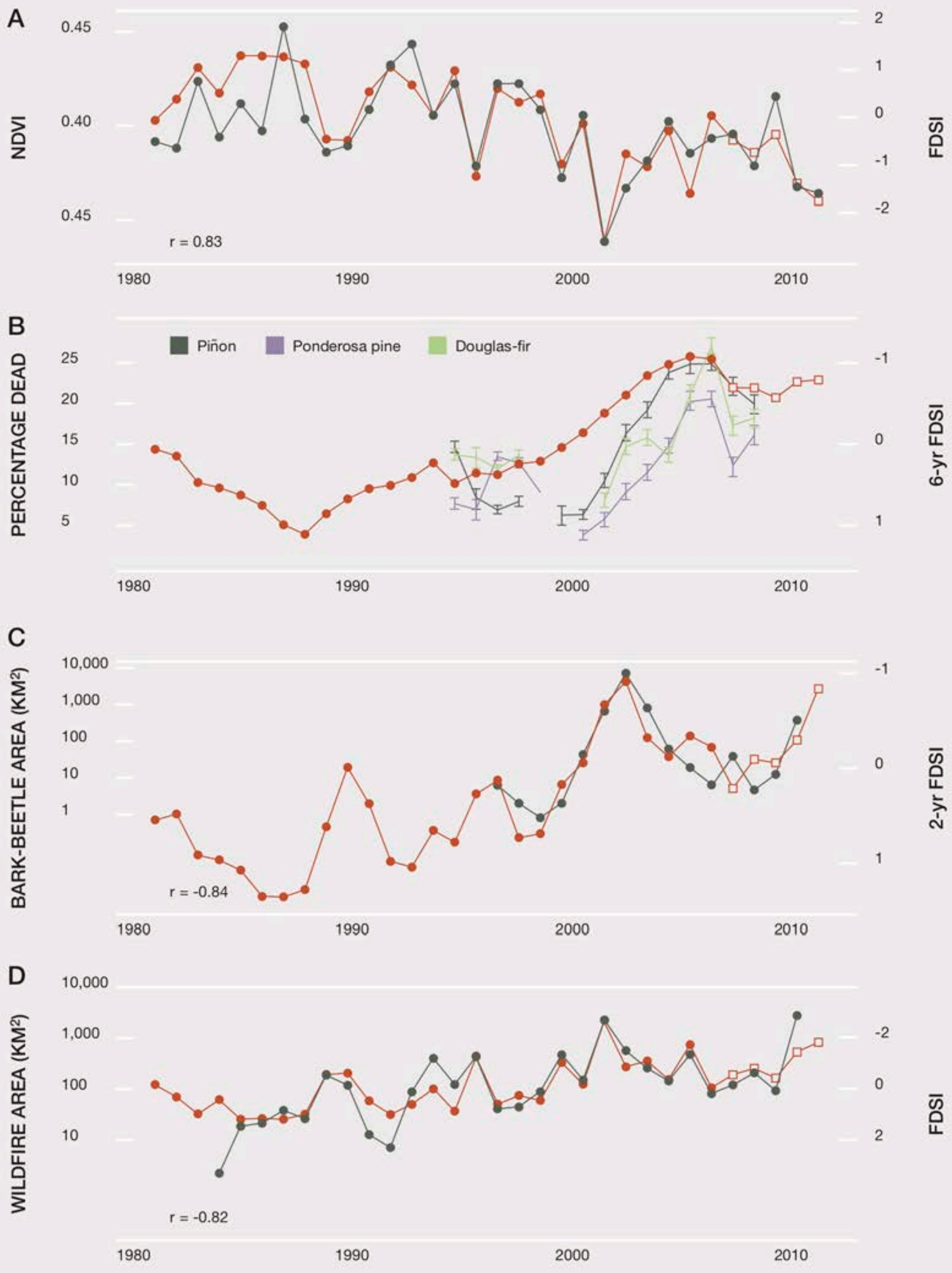
Recent trends. Moderate habitat degradation has occurred over the past 50 years, although forest cover is stable (Fig. 3.2, 3.2) (Hansen et al., 2013), and some sources report that the amount of forest cover has slightly increased (Keenan et al., 2015). Some 92% of the non-federal land in the USA that was in forest land use in 1982, remained as forest in 2007. Of the 12.8 million hectares of forest land that was transformed during this period, most (54%) was converted to developed lands; 22% went into pasture or rangeland, 14% changed to cropland or other another type of rural land, and about 10% went into water areas or federal ownership (USDA, 2007).

The arrival in recent decades of exotic pests and pathogens has caused declines in some of the most highly abundant tree species and genera in North America, including elms and hemlocks (Orwig et al., 2002) and ash and oaks (Juzwik et al., 2011). Tree mortality caused by insects and diseases was reported on nearly 1.82 million hectares in the USA in 2013 (USDA, 2015). Weed et al. (2013) identified 27 insects (6 non-indigenous) and 22 diseases (9 non-indigenous) that notably disturb North American forests. In Canada, the mountain pine beetle has killed trees on 20 million ha in British Columbia and Alberta. European earthworms, arriving in plant root balls and introduced for use as fishing bait, have invaded Canada and many parts of the USA and have caused population declines in many native understory herbaceous plant species (Holdsworth et al., 2007; Wiegmann & Waller, 2006). The worms feed on the upper layer of the forest soil, where symbiotic fungi occur, causing fungi as well as the plant species that host them to decline and leading to changes in soil properties, nutrient cycling and ecosystem functions (Frelich et al., 2006; Hendrix et al., 2006; Ewing et al., 2015; Hale et al., 2005; Resner et al., 2015). Oil extraction in the tar sands of Alberta has led to forest losses of 141,000 km^2 (Johnson & Miyanishi, 2008).

High-latitude forests in North America have warmed rapidly since the mid-1900s (Chapin et al., 2005; Allen et al., 2010). From 1902 to 2002 tree ring studies evidence declining growth, with increasing rates of decline since 1942, particularly in critical boreal conifer species (Lloyd & Bunn, 2007). The breeding ranges of some mobile species (e.g. certain bird species), have been expanding northward in association with climate amelioration (USDA, 2007). Current research suggests a northward shift of boreal forests is occurring (yet data is still limited) (Evans & Brown, 2017), with upward altitudinal shifts of tree species in some locations (Beckage et al., 2008).

In the southwestern part of the biome, over the past 30-40 years, forests have come under increasing stress as a result of severe drought. This has seen an increase in tree death, stronger outbreaks of bark beetle and an increase in the area affected by wildfire (Williams et al., 2013) (Figure 3.18), illustrating multiple effects and predicting future changes in forest composition.

Figure 3 18 Normalized Difference Vegetation Index (NDVI) (a), tree mortality (b), bark beetle outbreak (c) and area affected by fires (d) from 1980–2012 compared with the FDSI (Forest Drought Stress Index, red, right y-axis) for forests in the southwestern USA. Source: Williams *et al.* (2013).



South American subregion

Status. South American cool temperate forests are found in Chile and Argentina. Strongly isolated from the nearest closed-canopy forests on the eastern side of South America (Armesto et al., 1998), southern temperate forests are important for carbon sequestration and storage and play a pivotal role in water regulation (Armesto, 2009; Peri et al., 2012). In Chile, where most of southern temperate forest is found, around 78% of the original forest remains (calculated from Luebert & Pliscoff, 2006), thanks to large masses of remote forests in the southern part of the country, much of which is in protected areas. Several forest-dwelling mammals, nevertheless, are threatened (e.g. Darwin´s fox: *Pseudalopex fulvipes; huemul:*

Hippocamelus bisulcus), but overall southern temperate forest biodiversity in a far better state than in the Mediterranean-type climate forests to its north.

Plant species (including trees) richness in southern temperate forests is low (Table 3.4). Tree species richness drops off dramatically with latitude, while mean latitudinal range size increases (Arroyo et al., 1996). However, interestingly, these forests have higher woody phylogenetic diversity relative to their species richness than South American forests from lower latitudes (Rezende, 2017). Iconic organisms, including the smallest deer and one of the most long-lived tree species in the world, are important tourist attractions. Geographic isolation has fostered outstanding endemism levels across a wide array of taxa (Arroyo et al., 1996; Stattersfield et al., 1998; Villagrán & Hinojosa, 1997; Vuilleumier, 1985) and include a third of woody plant genera, two woody families (Arroyo et al., 1996), and almost all trees (Villagrán & Hinojosa, 1997); several endemics are shared with Mediterranean forest. Comprehensive surveys reveal large numbers of edible, medicinal, dye, basketry and ornamental plants and edible fungi in these forests used by indigenous peoples and local people (Tacón et al., 2006). The important ecosystem services supplied by southern temperate forests are enhanced by an especially high level of protection in the far southern part of their distribution (Luebert & Pliscoff, 2006).

Recent trends. Substantial habitat loss has occurred in the northern part of South American temperate forests over the past 40–50 years. The main losses came from deforestation for plantation forestry, farming, and raising of livestock. From the mid to late 90s until around 2013, 138,000 ha of native forest were lost in southern Chile, principally to plantation forestry (70%) (Instituto de Asuntos Públicos-Centro de Análisis de Políticas Públicas, 2016). From 1985 to 2011 a total gross loss of temperate forest of 30% was reported for the Coast Range in Chile but the net woody cover loss was only 5.1% due to other shrubland and agricultural and pasture land being converted to secondary forest following natural regeneration (Zamorano-Elgueta et al., 2015).

Twelve introduced mammalian herbivores (including three species of deer and beaver) are found in the southernmost forests, leading to altered forest regeneration and increased exotic plant richness in some forest types (Vázquez, 2002). Exotic plants are known to generate significant impacts on biodiversity of understory vascular plants, epigeal beetles and birds in *Nothofagus dombeyi* forest by diminishing species richness, abundance and diversity and generating modifications in assemblage composition (Paritsis & Aizen, 2008). The invasive *Ulex europeaus* has become a serious threat to Chilean agriculture and plantation forestry in some parts of the temperate forest zone (Norambuena et al., 2000). Exotic beavers cut down trees and have altered water regulation, silting levels and landscape values (see also chapter 4). Introduced conifers have now began to seed naturally in steppe vegetation and are associated with declines in plant species richness and cover (Taylor et al., 2016).

Fast-growing exotic plantation trees tend to consume more water than native trees and can be associated with reduced seasonal water provision (Lara et al., 2009). Nevertheless, there have been some recent positive signs of native forest recuperation. Between 1983 to 2007, in a part of the Araucania in Chile, the dominant land cover transitioned from agriculture to native vegetation, with largest increases occurring around residential areas found close to closed stands of native forest (Petitpas et al., 2016). These positive changes are attributed to the growth of tourism and a growing cultural preference for “natural” spaces.

On a longer timescale, in northwestern Patagonia in Argentina, during the last century, forests (mainly *Nothofagus*) expanded to cover almost 50% of the historically burned land, and more than 60% of the shrublands (Gowda et al., 2012). The estimated carbon stock recovery time for severely burned *Nothofagus* forests in Patagonia is 150–180 years (Bertolin et al., 2015) indicating a severe ecosystem service loss due to burning. However, regrowth is far from homogeneous in time and space: net forest expansion took place mainly from 1914 to 1973, probably favored by a wetter climatic period, and has shown a marginal retraction since then. Although forest gains remained high during the last 30 years, substantial areas of forests in this area were converted to grasslands and shrublands as a result of recent fires associated with extremely dry springs (Gowda et al., 2012). A major drought in 1998–1999 coincident with a very hot summer led to extensive dieback in a *Nothofagus* species (Suarez et al., 2004). In another dominant *Nothofagus* species, several periodic droughts have triggered forest decline as of the 1940s (Rodríguez-Catón et al., 2016).

Over the past 20-30 years, the biodiversity of southern temperate forests has become widely recognized for its ecotourism and tourism values. For example, the recent scientific finding of outstanding bryophyte diversity in the southern temperate forests, which led to the concepts of “miniature forests” and “tourism with a hand lens” (Rozzi et al., 2008) in the Cape Horn Biosphere Reserve on the southern tip of the continent, has seen a substantial increases in visitors, favoring local human well-being in an area where climate precludes agriculture and plantation forestry.

3.4.1.4 Mediterranean forests, woodlands and scrub

North American subregion

Status. The Mediterranean climate zone in North America encompasses the California Floristic Province, including southwestern Oregon, California west of the Sierra Nevada and a portion of northwestern Baja California, Mexico (Baldwin et al., 2012; see Ackerly et al., 2014, for a stricter definition and mapping of Mediterranean-climate regions based on current climate). The broader Mediterranean forests, woodland and scrub area has a very rich and endemic flora (Table 3.4) (Burge et al., 2016), with many evolutionary lineages represented (Baldwin, 2014). High levels of plant endemism are found in ephemeral vernal pools (Keeley & Zedler, 1998) and on serpentine soils (Anacker, 2014). California has more than 300 endangered and threatened species listed by the USA government, the largest for any USA state, and more than 100 others are listed by the state (California Natural Resources Agency, 2015). Hobbs & Mooney (1998) report 49 extinct taxa for seven groups of organisms (including some subspecies) (34 for plants) along with numerous cases of local population extinctions. According to the most recent account, 17 taxa (13 species and four subspecific taxa) of Californian vascular plants are globally extinct (Rejmánek, 2017) with 15 additional species extinct in California but found elsewhere (together 0.53% of the Californian flora); extinctions are associated with small range sizes and lowland habitats.

North American Mediterranean forests, woodland and scrub houses 991 species of alien plants and 109 species of alien vertebrates (including 26 mammals) (Zavaleta et al., 2016). Some 183 plant species are currently listed as invasive plants capable of damaging the environment and economy by the California Department of Food and Agriculture (California Natural Resources Agency, 2015). Coastal sage is very heavily invaded (Cleland et al., 2016). Brooms and gorse invade woodlands and shrublands and can displace native vegetation when not controlled (California Invasive Plant Council, 2017).

Forests in the Sierra Nevada play a critical role in water supply. Most urban and agricultural water originates in these mountains, and 30% of California’s water is stored for a part of the year in the snowpack. Healthy forests reduce flood risks and lead to more predictable water flows.

Recent trends. In the past 50 years, urbanization, exurban development, and agriculture have caused considerable conversion of natural habitat (Brown et al., 2005; Wilson et al., 2016); for example, a fourfold increase in vineyard acreage between 1976 and 2010 removed much oak woodland in coastal counties (Davis et al., 2016). Vegetation fragmentation — possibly exacerbated by changing climate in some cases – and the secondary effects of urbanization such as predation by urban cats on birds have reduced butterfly richness, bird abundances, genetic connectivity and species diversity in some taxa and produced declines in plant species richness in different vegetation types (Benson et al., 2016; Casner et al., 2014; Cooper et al., 2012; Johnson & Karels, 2016). Nevertheless, urban and semi-urban areas can house considerable plant diversity (Schwartz et al., 2006) and support high levels of bee diversity (Frankie et al., 2009) and thus could turn out to be very relevant for conservation.

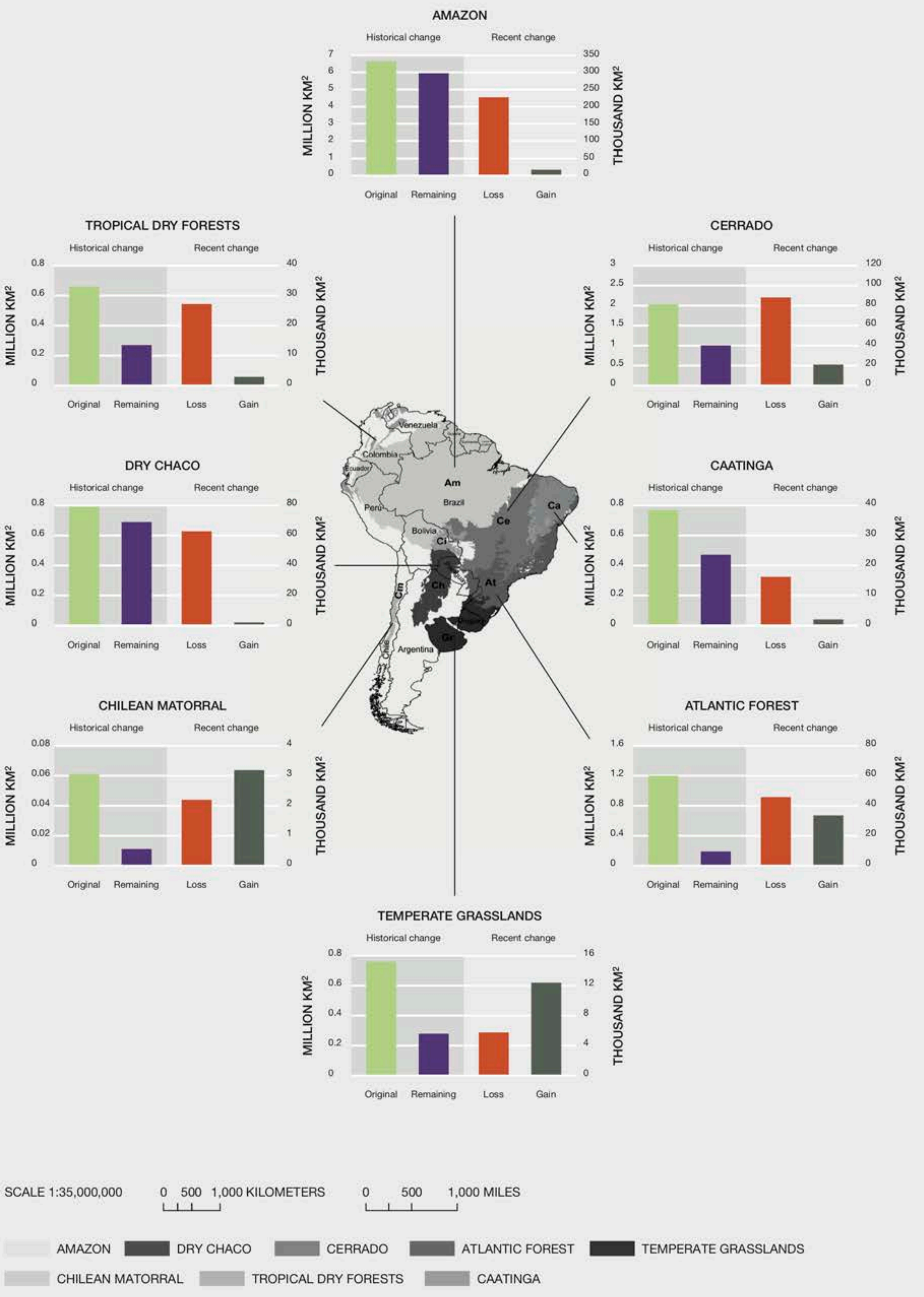
Mediterranean forests, woodland and scrub has experienced warming (Diffenbaugh et al., 2015). Upward elevational range shifts, consistent with warming, have been reported in small mammals (Moritz et al., 2008), birds (Tingley et al., 2009) and plants (Wolf et al., 2016), as well as earlier butterfly appearance (Forister & Shapiro, 2003) and arrival of migratory birds. Downward elevational shifts have also been reported in birds (Tingley et al., 2009) and plants (Crimmins et al., 2011). For plants, there is disagreement both about the trends and inferred link to climate (Stephenson & Das, 2011). Since the 1920s, tree densities

increased and size class distributions have changed in forests across California (Dolanc, et al., 2014; Dolanc et al., 2014; (McIntyre et al., 2015), in part due to changing fire regimes (see below). Reductions in the density of large trees are correlated with increased severity of summer water deficits (McIntyre et al., 2015).
California experienced a severe drought from 2012 to 2016, and even before it ended some calculations estimated that it exceeded in duration and intensity those observed for at least a century and possibly more than 1,000 years (Griffin & Anchukaitis, 2014). By one estimate, the intensity of the drought was increased by up to 27% due to increased temperatures on top of low rainfall (Williams et al., 2015). Widespread tree mortality has been observed, especially in Sierra Nevada conifer forests, with estimates exceeding 100 million dead trees spread over more than 3 million ha of forest (US Forest Service, 2016).

Several recent invasions of pathogens and disease have impacted biodiversity. Virulent pathogens affecting amphibians have been detected in a high proportion of wetlands (Hoverman et al., 2012); chytrid fungus has been attributed to amphibian declines in northern California, especially in high elevation populations of mountain yellow-legged frog (Piovia-Scott et al., 2015; Briggs et al., 2005). Sudden oak death (*Phytophthora ramorum*) arrived in the mid-1990s on horticultural trade plants and has caused extensive oak mortality in moist-climate, coastal woodlands (Zavaleta et al., 2016).

In Sierra Nevada pine forests, fire suppression led to marked increases in overall forest density, especially in small trees (McIntyre et al., 2015). Dense forests contribute to catastrophic wildfires that exceed the range of historical fire variability, such as the 104,000 ha Rim Fire in 2013, the largest fire on record in the Sierra Nevada (Kane et al., 2015). At mid- to high elevations, larger areas are being burned, likely due to past fire suppression, changing fire management policies, and warmer and drier climatic conditions. The length of the fire season increased by over two months from 1970 to 2003, associated with warming trends (Westerling et al., 2006). More frequent fire has led to much type conversion of shrubland to grassland (Zedler, 1995; Halsey & Syphard, 2016).

Figure 3 19 **Total change in vegetation type and recent change (2000–2012) in forest cover for several biomes in South America. Source: Modified from Salazar *et al.* (2015).**



South American subregion

Status. Part of a Biodiversity Hotspot (Myers et al., 2000), South American Mediterranean forests, woodland and scrub found in central Chile, under a broad definition, is characterized by high endemism, richness and phylogenetic diversity (Arroyo et al., 2002; Rundel et al., 2016; Scherson et al., 2017) (Table 3.4). Around 50% of Mediterranean forests, woodland and scrub has been transformed (Luebert & Pliscoff, 2006) – this percentage is considerably higher under a narrower definition of the biome (Figure 3.19). Many native species are threatened (Ministerio del Medio Ambiente, 2017), although only a small fraction (ca. 3.5%) of all Chilean species have been analyzed (OECD/ECLAC, 2016). Alien species including close to 600 plant species (Fuentes et al., 2015; Jiménez et al., 2008), >100 insect species (Ministerio del Medio Ambiente, 2017), and 30 vertebrate species (Iriarte et al., 2005; Jaksic, 1998) – several of which are considered harmful by stakeholders (COCEI, 2014) – are abundant in disturbed areas, urban areas, and semi-natural grasslands (Arroyo et al., 2000; Contreras et al., 2011; Estay, 2016; Figueroa et al., 2011; Gärtner et al., 2015; Jaksic, 1998; Martín-Forés et al., 2015). Plant-animal interactions for pollination and seed dispersal are especially well developed and critical for vegetation regrowth and restoration. Other biodiversity-NCP links include the provision of medicinal plants (Niemeyer, 1995), nectar and pollen sources for honey making (Bridi & Montenegro, 2017), runoff control on steep slopes (Pizarro Tapia et al., 2006) and the aesthetic value of the rural-natural landscape mosaic.

Recent trends. One study suggests Mediterranean forests have recently increased but this is acknowledged as likely due to the inclusion of exotic forests (Figure 3.19) (Salazar et al., 2015). National data for approximately between the last decade of the past century and the first of this century for Mediterranean-climate forest (V-VIII Regions) come up with a net loss of 99,451 ha, mainly distributed among conversion to exotic plantation forestry (24%), agriculture (11%), scrub and open vegetation (59%), and urban areas (2%) (Instituto de Asuntos Públicos-Centro de Análisis de Políticas Públicas, 2016). Exotic plantation forestry accounted for most of the forest loss in the southern part of the biome. Although some passive renovation has been occurring, previously forested areas tend to remain as scrub (Schulz et al., 2010, see also Hernández et al., 2016). Plantation forests have been shown to have a negative effect on annual stream flow in the biome (Iroumé & Palacios, 2013) and loss and fragmentation of native forests have negatively affected many plant and animal species (Braun & Koch, 2016; Bustamante & Castor, 1998; Muñoz-Concha et al., 2015; Saavedra & Simonetti, 2005; Soto-Azat et al., 2013; Vergara et al., 2013; Vergara & Simonetti, 2004) and pollination services to native plants (Valdivia et al., 2006).

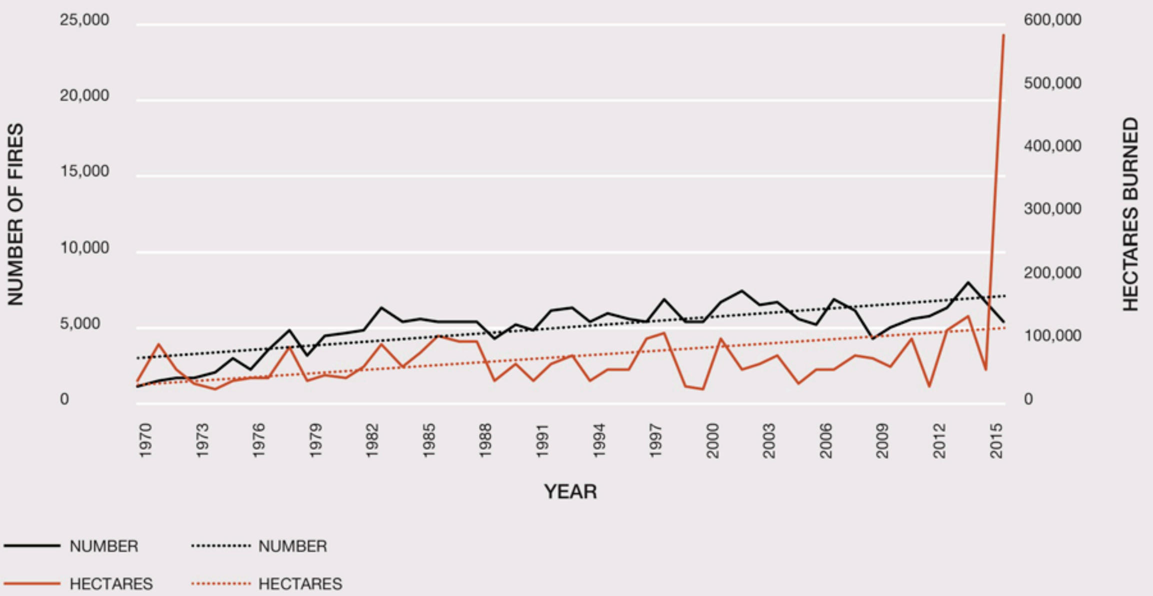
Among the new insect invaders (Grez et al., 2010; Ide et al., 2011; Lanfranco & Dungey, 2001; Montalva et al., 2011) and introduced fungal diseases (Durán et al., 2008; Slippers et al., 2009), some are spreading at remarkable rates (e.g. Schmid-Hempel et al., 2014, Grez et al., 2016). *Bombus terrestris*, introduced in the 1990s for crop pollination, moved rapidly into Argentina and is now displacing native bumblebees there (Geslin & Morales, 2015). Many native plant species in Mediterranean forests, woodland and scrub are visited by *B. terrestris* (Montalva et al., 2011), but the impacts of *B. terrestris* on the wider bee fauna of central Chile (Table 3.4), likely to assist crop pollination, are unknown. The escaped introduced frog *Xenopus laevis* has now been found to harbor amphibian pathogens, posing a potential threat to the biome´s highly endemic amphibians (Soto-Azat et al., 2016) and showing that single invasions can have secondary effects.

Between 1994 and 2015, fire affected close to 128,000 ha of closed Mediterranean forest as well as huge areas of exotic plantation forests (based on Instituto de Asuntos Públicos-Centro de Análisis de Políticas Públicas, 2016). A recent megadrought ushered in a notable increase in fire frequency and extent in Chile (with most fires in the Mediterranean area) (Figure 3.20), culminating in the massive forest fires of the austral summer of 2016 which affected 518,000 ha, including 105,000 ha of native forest and 284,000 ha of exotic forest plantations (CONAF, 2017), mostly in the Mediterranean zone. Although there is still some discussion on the issue, it is generally agreed that unlike North American Mediterranean forests, woodland and scrub, South American Mediterranean forests, woodland and scrub was cut off from natural lightning strike fires as of the Miocene and consequently is not strongly adapted to fire (Rundel et al., 2016). Although many native woody species can resprout after fire, recovery of Mediterranean forest may require 25–30 years and often is never complete (Montenegro et al., 2003), indicating limited resilience. Fire additionally provokes the entrance of invasive species (Contreras et al., 2011; Gómez-González et al., 2011; Gómez-

González & Cavieres, 2009; Pauchard et al., 2008) further altering species composition and NCP delivery. Warmer and drier conditions in central Chile also saw a significant decrease in growth rates of *Nothofagus macrocarpa* as of the 1980s (Venegas-González et al., 2018).

Figure 3 20 **The number of forest fires (grey line) and hectares affected (red line) in Chile between 1970 and 2016.**

Year refers to the austral spring-summer season beginning in the year indicated. Source: Data from http://www.conaf.cl/incendios-forestales/incendios-forestales-en-chile/estadisticas-historicas/.



Urban expansion in central Chile, often recent, has also contributed to local habitat and species losses (Pauchard et al., 2006; Pavez et al., 2010; Simonetti & Lazo, 1994). However, urban spaces clearly can play an important role in maintaining biodiversity, as seen by the 42 native bee species in a semi-natural botanical garden in Santiago (Montalva et al., 2010).

3.4.1.5 Tundra and high mountain habitats

North American subregion

Status. Species richness beyond latitudinal treeline in North American arctic tundra is low in relation to its vast area (Table 3.4), and decreases with increasing latitude (Meltofte, 2013; Walker, 1995). Endemism is rare because many tundra-adapted taxa are distributed across both North American Arctic tundra (including Greenland) and the Eurasian arctic tundra. For example, 80% of vascular plant species in the Arctic are common to both regions, so just 1.1% of North American Arctic tundra vascular plant species are endemic (Callaghan et al., 2004; Elven at al., 2011). The extent and biodiversity of the North American arctic tundra remain largely unchanged compared to pre-European settlement, with localized reductions in extent associated with natural resource extraction and permanently settled villages and cities (Raynolds et al., 2014; Young & Chapin III, 1995). Non-native species in arctic tundra are uncommon and usually associated with human activity (Ackerman & Breen, 2016; Elsner & Jorgenson, 2009; Forbes & Jefferies, 1999). Carbon storage in North American Arctic tundra soils is high relative to other biomes, due to low rates of organic matter decomposition. Hugelius et al. (2013) estimate 25–100 kg C/m^2 across most of North American Arctic tundra. At local scales, stocks of carbon and soil nutrients vary widely based on vegetation community type (Shaver et al., 2014). Across all community types, soil nitrogen is dominated by non-mineral forms, so primary productivity in North American Arctic tundra is often limited by rates of nitrogen mineralization (Chapin & Shaver, 1985; Shaver et al., 2014; Chapin et al., 1995).

Globally, North American Arctic tundra stores carbon in soils frozen year-round called permafrost (Michaelson et al., 1996). Biodiversity alters this ecosystem service through plant traits (Chapin et al., 2000). For example, plants with extensive mat growth forms, like *Sphagnum* spp., insulate permafrost soils from direct sunlight (O'Donnell et al., 2009). Permafrost stores 1,330–1,580 picograms of organic carbon, nearly half of the global organic carbon pool (Schuur et al., 2015). Locally, North American Arctic tundra benefits subsistence hunters, providing game species including caribou (*Rangifer tarandus*) and ptarmigan (*Lagopus* spp.) (Alaska Department of Fish and Game, 2016).

Western North American alpine ecoregions contain diverse ecosystems and over 1,400 plant species (Malanson et al., 2015). Similarity among plant communities throughout mountain ranges of western North America is driven primarily by geographic distance, but also by hydroclimatic variables (Malanson et al., 2015). Endemism is common in western North America (45% of plant species), while exotic species are rare (Malanson et al., 2015). Native biodiversity of the western North America high altitude areas remains largely intact since European colonization. The eastern North America alpine ecoregion (Appalachian Mountains) is understudied and lacks a comprehensive record of biodiversity.

Recent trends. Species richness has not changed significantly in North American Arctic tundra. Some boreal plant species, including trembling aspen (*Populus tremuloides*) and white spruce (*Picea glauca*), have expanded locally into North American Arctic tundra due to infrastructure development (Ackerman & Breen, 2016; Elsner & Jorgenson, 2009). The only reported extinction is the Eskimo curlew (*Numenius borealis*), an over-exploited migratory shorebird (Harris et al., 2012). It is very well established that woody deciduous shrubs native to North American Arctic tundra have become increasingly dominant due to warming (Fraser et al., 2014; Moffat et al., 2016; Myers-Smith et al., 2011, 2015; Naito & Cairns, 2015; Pizano et al., 2014; Sturm et al., 2001; Tape et al., 2006; Tremblay et al., 2012). While the overall area of North American Arctic tundra has not changed significantly, habitat has been degraded biome-wide due to high-latitude concentration of atmospheric pollutants (Hung et al., 2010; Krachler et al., 2005; Quinn et al., 2007) and regionally due to road construction (Auerbach et al., 1997; Hinkel et al., 2017; Raynolds et al., 2014; Walker & Everett, 1987).

Above-ground standing biomass has increased at low latitudes in the Arctic (Epstein et al., 2012), and vegetation cover has increased in mid- to high-latitudes, possibly due to increased maritime climate moderation linked with sea ice decline (Bhatt et al., 2010). Despite elevated productivity, overall carbon storage across North American Arctic tundra has decreased since 1970 due to warming-induced carbon losses from soil (Hayes et al., 2014; Hinzman et al., 2005; Oechel et al., 2000; Schuur et al., 2009). Recent trends in water balance are uncertain, though there has been a general acceleration of the hydrologic cycle across North American Arctic tundra due to changes in precipitation, evapotranspiration, and drainage conditions (Andresen & Lougheed, 2015; Bring et al., 2016; Cherry et al., 2014; Hinzman et al., 2005; Liljedahl et al., 2016; Oechel et al., 2000; Rawlins et al., 2010; Vihma et al., 2016; Young et al., 2015).

Greater variability in the timing and magnitude of precipitation events in North American Arctic tundra has decreased accessibility and yield for subsistence hunters (Berkes & Jolly, 2002; Rennert et al., 2009). Further, atmospheric deposition of pollutants in North American Arctic tundra has threatened the health of local communities through the bioaccumulation of toxins in organisms used for food (Kelly & Gobas, 2001). To improve community resilience to these changes, Chapin et al., (2006) suggest diversifying the economies of indigenous communities by reinvesting tax revenue from natural resource extraction into local education and infrastructure.

The extent of alpine habitat in western North America has decreased due to warming-induced treeline advance, though rates of advance are spatially variable (Elliott, 2011; Harsch et al., 2009). Some degradation from logging, pasturing, and recreation is evident, but these disturbances have been minor compared to in alpine zones on other continents (Bowman & Seastedt, 2001). Recent changes include increased shrub cover and diminished species richness, likely in response to a combination of climatic change, and high levels of nitrogen deposition from anthropogenic pollution (Elmendorf et al., 2012; Formica et al., 2014; Sproull et al., 2015). The most notable change among alpine fauna populations is the rapid decline of the

American pika, a small alpine mammal experiencing an upslope range contraction in response to climate warming (Beever et al., 2011, 2016; Stewart et al., 2015).

South American subregion

Status. South American high elevation habitats occur principally along the entire length of the Andes (Arroyo & Cavieres, 2013). These habitats, found under a variety of climatic conditions, are remarkably rich in plant species (Table 3.4.) and evolutionary lineages (Sklenář et al., 2011) and support the richest tropical alpine flora in the world (Sklenář et al., 2014). High-elevation habitats support many species of large mammals (Ojeda et al., 2003), lizards (Pincheira-Donoso et al., 2015), birds (Arbeláez-Cortés et al., 2011; Fjeldså & Rahbek, 2006; Fjeldså, 2002) and pollinating insects (Arroyo et al., 1982). Puna lakes supports 58 species of native fishes (Vila et al., 2007), diverse waterfowl (Cendrero et al., 1993), and rich assemblages of gastropods (Kroll et al., 2012), while hot springs and periglacial soils fascinating assemblages of microorganisms (Costello et al., 2009; Schmidt et al., 2009).

Species-level endemism and turnover in the high tropical Andes can be very high (Londoño et al., 2014; Sklenář et al., 2014). Mountain-top vegetation is richer in plant genera and species in páramo compared to puna (Cuesta et al., 2017) but the puna and southern Andean steppe house more endemic genera than páramo (Arroyo & Cavieres, 2013). Páramo and puna have long been under human influence (Box 3.6), but more intensely so as of colonial times (Vásquez, et al., 2015). In the high southern Andes, human influence has never been very great. Today it is principally via low-intensity transhumance summer grazing, skiing, and mining. Some Andean threatened species rely heavily or partially on páramo, among them the Andean condor (*Vultur gryphus*), the mountain tapir (*Tapirus pinchaque*), the Andean bear (*Tremarctos ornatus*), and several deer species (*Pudu mephistophiles, Mazama rufina, M. americana and Odocoileus virginianus*) (Muñoz et al., 2000). In general, South American high elevation habitats have garnered few alien plant species (Alexander et al., 2016; Barros & Pickering, 2014; Luteyn, 1999; Urbina & Benavides, 2015) and these are mostly confined to disturbed areas. A few serious recent invasions have now been recorded for páramo, as for example *Ulex europaeus* in Colombian páramos (see Table 3.2) and more can be expected in the future given trends in alteration (Box 3.6).

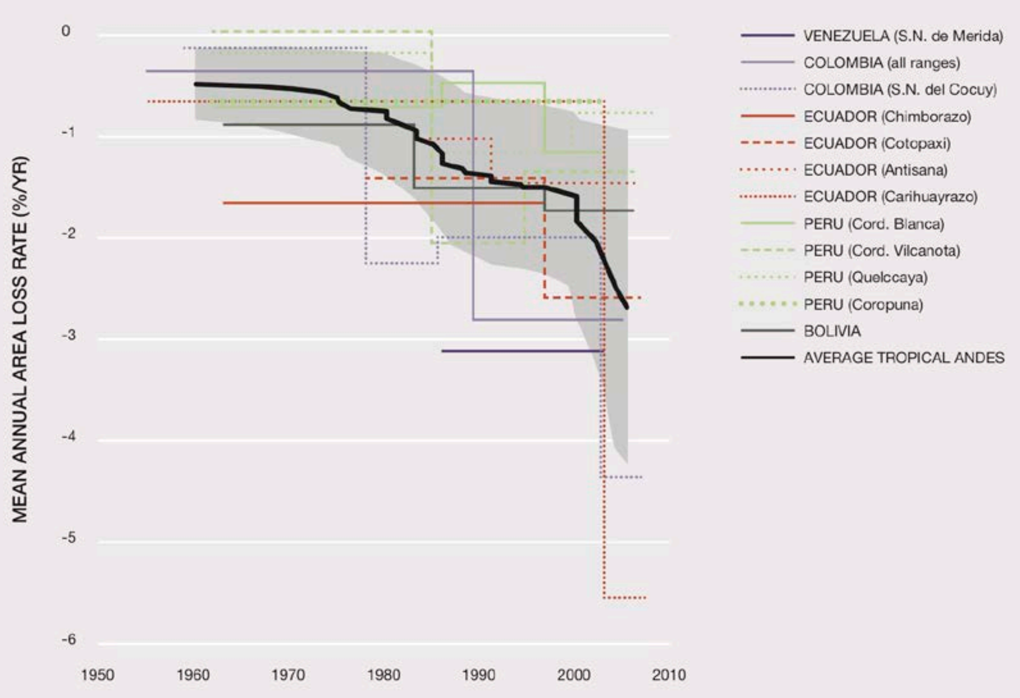
Páramo and wet puna are notable for rapid water absorption but slow water release (Buytaert et al., 2005), which is important for the support of agriculture and the delivery of water to lowland areas. For example, 60% of water in Colombia derives from páramo (Cadena-Vargas & Sarmiento, 2016). Carbon storage is páramo is important (Forero et al., 2015). In particular, it is very high in páramo peatlands (Hribljan et al., 2015; 2016). Soils under older pine plantations in páramo have lower carbon content and retain less water compared with natural grasslands (Farley et al., 2004, 2005, 2013) and the loss of water retention after afforestation may be the dominant factor in carbon loss (Farley et al., 2004).

Recent trends. Páramo and puna have seen an increasing trend for afforestation with fast-growing exotic trees and intensive agriculture. Both afforestation and cultivation have been found to increase streamflow variability and decrease catchment regulation capacity and water yield.(Ochoa-Tocachi et al., 2016). Moreover, shifts to agriculture lead to a loss of microbial functional diversity in páramo, which is reflected in lower metabolic activity. Fishing, based on native species, is a longstanding tradition in some large high Andean lakes. However, the introduction of trout (and silversides) together with more invasive fishing techniques has seen a decline in endemic native fish (Vila et al., 2007).

Box 3.6. The role of páramo and puna for human well-being.

Humans were living at 4,480 m.a.s.l some 11,500 years ago in the puna of Peru (Rademaker et al., 2014) and at 3,000–3,600 m.a.s.l some 13,000 calibrated years before the present on the Chilean altiplano (Núñez et al., 2002). High altitude indigenous peoples of the páramo and puna have accumulated a wealth of ILK on high Andean biodiversity, especially useful plants (Aldunate et al., 1983; Brandt et al., 2013; Califano & Echazú, 2013; Huamantupa et al., 2011; Monigatti et al., 2013; Pauro et al., 2011; Ramos et al., 2013; Thomas et al., 2008, Villagrán et al., 2003) and have developed resilience to climatic extremes by managing alternative crop varieties Local inhabitants have developed their own taxonomic systems reflecting thousands of years of interchange between different linguistic groups (Aldunate et al., 1983; Villagrán & Castro, 2003). High Andean bogs in the arid puna are key habitats for native camelids which sustain the livelihoods of high altitude peoples (Borgnia et al., 2008; Tirado et al., 2016). The integrity of the páramo and puna thus is critical to conserving ILK and for the livelihoods of local inhabitants. Páramo and wet puna play critical roles in supplying water supply to lowland Andean areas. Water availability is threatened on two counts. First, severe glacier dieback has occurred over the past decades (Figure 3.21). Second, páramo and puna are increasingly being converted to other land use types involving higher water-demanding trees (c.f., Hofstede et al., 2002) and crops. Around 16% of Colombian páramos have been now been transformed (Bello et al., 2014) mainly due to cropping and pastures. Peruvian Jalca grasslands were transformed at the rate of 1.5% per year over a 20 year period starting 1987 due mostly to more intensive agriculture and afforestation (Tovar et al., 2013). Rapid glaciar melt also portends landslides on unconsolided deglaciated substrates following heavy rains.

Figure 3 21 **Compilation of mean annual area loss rates for different time periods for glaciated areas in the northern Andes between Venezuela and Bolivia. Source: Rabatel *et al.* (2013).**



High elevation areas have warmed in the southern (Falvey & Garreaud, 2009) and northern (Hofstede et al., 2014) Andes. Whether and the degree to which anthropogenic warming has affected tree growth and the position of the treeline along the Andes are still somewhat unclear. Anthropogenic warming seems to have affected tree growth and increased recruitment above treeline in some places, but not in others

(Aravena et al., 2002; Daniels & Veblen, 2004; Fajardo & McIntire, 2012; Lutz et al., 2013; Rehm & Feeley, 2013; Villalba et al., 1997). Some tree species have been moving upward below treeline (Feeley et al., 2011). Lack of or very slow upward movement of the treeline might reflect recruitment difficulties in high altitude grasslands (Rehm & Feeley, 2013, 2015, 2016) or under reduced precipitation in some parts of the Andes. Historical comparisons suggest upward movement in some plant and beetle species in the northern Andes (Moret et al., 2016; Morueta-Holme et al., 2015; but see Sklenář, 2016). In the longer term, contractions of total area occupied can be expected in high elevation species under warming given that land area decreases with increasing elevation throughout much of the Andes. Warmer soil conditions in the páramo are expected to cause faster organic carbon turnover thereby decreasing below-ground organic carbon storage (Buytaert et al., 2011).

3.4.1.6 Tropical savannas and grasslands

South American subregion

Status. In South America, this biome occurs mainly in Brazil, Paraguay, Argentina, Venezuela, Colombia, and Bolivia. The largest extents are the Cerrado, originally covering around 2 million km^2, and the Dry Chaco, originally over ¾ of a million square kilometers (Salazar et al., 2015).
Comprising a mosaic of tall savanna woodlands, gallery forests and treeless grasslands, Cerrado is a recognized Biodiversity Hotspot (Myers et al., 2000). It is characterized by high plant and bird species richness and endemism (Table 3.4). Birds use many habitats, especially forested areas (Carmignotto et al., 2012), lizards prefer open interfluvial habitats (Nogueira et al., 2009), while large mammals use a wide range of habitats (Lyra-Jorge et al., 2008), including converted land (Cárceres et al., 2010). Over half of Cerrado mammals and birds consume fruits with about one-third of Cerrado plants depending on birds and mammals for fruit and seed dispersal (Kuhlmann & Ribeiro, 2016). Mammals and birds thus are fundamental for natural Cerrado regeneration.

Some 52% of all South American Cerrado has been converted (Salazar et al., 2015) (Figure 3.19). According to (Beuchle et al., 2015), 47% of Brazilian Cerrado has been transformed. Remaining Cerrado is highly fragmented with the landscape dominated by crops and pastures (Carvalho et al., 2009). Fragmentation reduces species richness and alters the composition of small mammals land (Cáceres et al., 2010), and birds (Marini, 2001). However, large mammals, which tend to use the entire the landscape, appear less susceptible (Cáceres et al., 2010; Vynne et al., 2014). Shrubby pastures in Cerrado hold far more bird species than cleared ones and obligate natural grassland bird species do not adapt well to pastures (Tubelis & Cavalcanti, 2000). Butterfly richness and beta diversity are lower in disturbed riparian Cerrado forest (Cabette et al., 2017). Additional threats to Cerrado biodiversity are fire suppression (Durigan & Ratter, 2016) and woody encroachment (Stevens et al., 2017). Cerrado is resilient to fire, expressed in rapid post-fire recuperation and fire aids in maintaining the mosaic structure of Cerrado. Replacement of grassy savannas with forests is also considered a threat (Veldman et al., 2015) because dense tree cover severely limits the richness and productivity of light-demanding herbaceous plants while reducing habitat for animals adapted to open environments. Several African grasses which were introduced into Brazil for cattle grazing are now highly invasive in the Cerrado leading to reductions in native plant species (Almeida-Neto et al., 2010). In the phosphorus-poor Cerrado, the addition of phosphorus tends to increase the biomass of alien C4 grasses (Lannes et al., 2016).

Some 34% of dry Chaco habitat has been converted (Figure 3.19) (Salazar et al., 2015). The Gran Chaco has a long history of colonization and land use change, beginning with subsistence hunting by native people. Over the past 200 years, dry Chaco has experienced drastic land use changes as a result of intensive agriculture, livestock production and logging (Eva et al., 2004; Hoyos et al., 2013). Moreover, deforestation and the introduction of domestic cattle have led to the elimination of fire-climax grasslands and altered forest composition and structure (Bucher, 1982; Gasparri & Grau, 2009). Chaco conversion has had negative effects on biodiversity. Almost 50% of the largest frugivorous mammals and 80% of the largest herbivores in the Argentine Chaco are threatened and exhibit declining populations; this is expected to change vegetation composition since more than half of Chacoan woody plant species display endozoochory as their seed dispersal mechanism (Periago et al., 2015).

Recent trends. The South American tropical and subtropical savannas and grasslands assessed here are strongly imperiled. As of around the 1970s, pasture development for cattle grazing and extensive and mechanized agriculture intensified, leading to the transformation of Cerrado into a vast commercial production landscape with concomitant charcoal production for the steel industry. Brazilian Cerrado suffered a gross loss of around 266,000 km^2 of natural vegetation between 1990 and 2010, but with a significant amount of regrowth also occurring (Beuchle et al., 2015). Although the annual net rate of loss (total loss adjusted for regrowth) slowed in the last decade of the past century, overall conversion occurred an average annual net rate of –0.6% between 1990 and 2010 (Beuchle et al., 2015). Between 2003 and 2013, the northeast agricultural frontier in Brazil more than doubled from 1.2 to 2.5 million ha, with 74% of new croplands sourced from intact Cerrado (Spera et al., 2016). Shifts from Cerrado to cultivation have resulted in huge soil losses under erosive storms (12.4 t/ha/yr for bare soil compared to 0.1 t/ha/yr for Cerrado) (Oliveira et al., 2015).The Paraná river basin suffered a 66% decrease in forest cover between 1977 and 2008, with a 3.5% annual rate of forest loss (Bianchi & Haig, 2013). A recent review (Hunke et al., 2015) concluded that while conversion of Cerrado did not alter total soil nitrogen, nitrogen enrichment in agricultural catchments has increased, indicating fertilizer impacts and potential susceptibility to eutrophication; moreover, pesticides are consistently found throughout the entire aquatic system. Part of the loss of woody cover in the Cerrado is due to charcoal production (Ratter et al., 1997). For example, 34.5% of around 5.5 milion tons of charcoal produced in the Brazil in 2005 still came from native Cerrado species in spite of efforts to transition to planted forests (Duboc et al., 2007).

Like Cerrado, the Chaco has recently undergone extensive transformation (c.f., Figure 3.19). Rapid loss of chacoan dry forest has been documented in Bolivia, Paraguay and Argentina (Gasparri & Grau, 2009; Grau et al., 2005; Zak et al., 2004), mostly due to agriculture (mainly, soybean). For the Cordoba area in Argentina, Zak et al. (2004) estimated clearing of 1.2 million ha between 1969 and 1999. For North West Argentina between 1972 and 2007, another 1.4 million ha were removed (Gasparri & Grau, 2009). According to Fehlenberg et al. (2017), some 7.8 million ha out of a total of 110 million ha of dry Chaco in all countries was converted between 2000-2012, (principally to support soybean production and cattle ranching).

Conversion of vegetation has facilitated the spread of invasive species, like *Pyracantha angustifolia* (Rosaceae), which is now widely spread in the Chaco Serrano of Argentina (Tecco et al., 2006). According to these authors, this species can potentially enhance the recruitment of forest species. However, a considerable number of other exotic woody species, and especially *Ligustrum*, are also favored by the presence of this exotic shrub (Tecco et al., 2006).

3.4.1.7 Temperate grasslands

North American subregion

Status. Grasslands were once widespread in midwestern North America, occurring in a mosaic of tallgrass prairie and savanna (Nuzzo, 1986). Prior to European settlement, the central prairie of North America is thought to have ranged from southern Alberta, Saskatchewan and Manitoba south to mid-Texas, and from the foothills of the Rocky Mountains eastward into Indiana, Kentucky and Ohio and southwestern Ontario, covering about 2.4 million km^2 (The Nature Conservancy, 2009; USDA & USDOI, 2012). Diverse grasslands are major reservoirs for belowground carbon storage and prevention of soil loss due to erosion. Grasslands also serve as buffers increasing ecosystem nutrient uptake reducing runoff of agricultural waste and fertilizer into water bodies. Declines have been greatest in the mixed-and tall-grass prairie, with estimates of less than 5% (Sampson and Knopf, 1994) to 0.5% (The Nature Conservancy, 2009; USDA & USDOI, 2012) of the pre-European settlement tall-grass prairie remaining. Currently, approximately 50% of the Great Plains - about 148 million hectares in total - remains in grassland (i.e., not in annual crops or developed land) (WWF, 2017a).

Grassland vegetation structure is strongly influenced by fire frequency, driven by topographic barriers to the spread of fire (rivers, lakes, and bluffs), with oak savannas and prairies occurring on sites exposed to frequent fire (Peterson & Reich, 2008). Prior to modern settlement, fires annually burned large areas of the tallgrass prairie biome of North America (Gleason, 1913). Most prairie and savanna ecosystems were plowed under for agricultural uses or succeeded to forest following reductions in fire frequency. Prairie and savanna ecosystems are now exceedingly rare and mostly restricted to sites with infertile sandy soils that were unattractive for agricultural uses or where succession to woodland was slow following reductions in fire frequency (Nuzzo, 1986; Peterson & Reich, 2001; Will-Wolf & Stearns, 1999).

Fire suppression and agricultural land uses are important causes of habitat and biodiversity loss. For example, after conversion of all but 0.1% of tallgrass prairie in the USA state of Iowa, recent surveys found only 55% (491) of the original plant species formally known to be present there (Smith, 1998; Wilsey et al., 2005). Fire suppression, exacerbated by fragmentation has caused a decline in small seeded and short stature species, as well as legumes, many of which are fire-dependent or require open areas (Leach & Givnish, 1996). In experimentally restored prairie/savanna systems, plant species richness and phylogenetic diversity are significantly higher in frequently burned grasslands than in unburned forests on the same soil conditions (Cavender-Bares & Reich, 2012; Peterson & Reich, 2008). Efforts to restore biodiversity and ecosystem services often fall short of the levels observed in remnant grasslands and other ecosystems (Benayas et al., 2009; Martin et al., 2005).

Bison were formerly dominant herbivores and a keystone species throughout the Great Plains (Knapp et al., 1999). During the mid-1800s bison were reduced from tens of millions to only a few thousand individuals, subsequently recovering to more than 100,000 individuals. Bison grazing maintains grassland plant diversity by suppressing dominant warm-season grasses that would otherwise out-compete many rare wildflowers (Collins, 1998). Many populations of other animals dependent on prairie systems, including mammals and birds, have declined or are now absent from large portions of their historical range.

Recent trends. In the Great Plains region, 21.4 million ha of grassland have been converted to cropland since 2009. This loss represents almost 13% of the 170 million ha that remained intact (i.e., not in annual crops) in 2009. The average annual rate of loss of grasslands was 2% between 2009 and 2015. In 2016, only 148 million ha of grassland remained intact in the Great Plains (Northern Great Plains Program, 2016; WWF, 2016). A report based on data from the USA and Canadian governments, indicates that more than 21 million ha of land in the Great Plains have been converted to cropland since 2009. From 2014 to 2015 alone, approximately 1.5 million ha were lost. Endemic grassland bird species have shown steeper declines than any other group of North American bird species (USGS, 2003). Since the 1960s, populations of four key species have declined by as much as 80% with annual declines as follows: McCown’s Longspur (*Rhynchophanes mccownii*), 6.2% decline per year; the chestnut-collared longspur (*Calcarius ornatus*), 4.4% decline per year; lark bunting (*Calamospiza melanocorys*) 4.1% decline per year and Sprague’s pipit (*Anthus spragueii*), 3.5% decline per year. The decline of these grassland species has been attributed directly to the loss of intact grasslands throughout the Great Plains region (Northern Great Plains Program, 2016; WWF, 2016). Loss of prairie plant diversity (Leach & Givnish, 1996; Wilsey et al., 2005) causes loss at higher trophic levels, including numerous insects and other organisms above- and belowground (Knops et al., 1999; Lind et al., 2015; Scherber et al., 2010; Siemann et al., 1998). Bees, important for pollination services, have declined; the rusty-patched Bumble Bee (*Bombus affinis*) which was declared an endangered species under the USA Endangered Species Act in 2017, once extended from the Dakotas and Nebraska, east across the Midwest and south to the Carolinas. Its population declined by 87% between 2011 and 2016. Other species that were once common in the Great Plains such as the western bumble bee (*Bombus occidentalis*) and the American bumble bee (*Bombus pensylvanicus*) are also declining severely (Northern Great Plains Program, 2016;WWF, 2016).

South American subregion

Status. This biome, as defined in the assessment, includes the Río de la Plata grasslands, Patagonian steppe and semi-desertic Monte vegetation, and thus includes many different vegetation types. Here, in our detailed analysis, we focus on the Río de la Plata grasslands, found principally in Argentina and Uruguay and extending into southern Brazil. These grasslands sustained grazing as of the 1600s. Fully 70% of the grasslands, formerly occupying an estimated ¾ of million square km, have been replaced (Salazar et al., 2015) by crops, pastures or afforestations. In Argentina, about one in every three plant species growing in natural or semi-natural pampas is non-native. Although there are still very few natural parks protecting the Río de la Plata grasslands, some recent efforts on grassland conservation have been notorious (e.g. Sistema Nacional de Áreas Protegidas-Uruguay, Alianza del Pastizal).

Recent trends. Profound changes, affecting key ecosystem functions and ultimately human well-being, have occurred in South American temperate grasslands. Livestock grazing for over 400 years has reduced soil organic carbon stocks by an estimated 22% (a reduction of 1.5 picograms of carbon for the whole region) and net primary production by 24% (Piñeiro et al., 2006). Cropping reduced soil organic carbon stocks by 20 to 30% in a few decades (Alvarez, 2001, 2005). Soil nutrients have been also depleted in croplands (near 30% of soil nitrogen and 80% of soil phosphorus) and rangelands (19% of soil nitrogen. Nutrient losses have triggered large increases in fertilizer use with beneficial effects for food production, but detrimental effects on air and water pollution (Portela et al., 2006, 2009). Crop rotation with pasture in the past helped maintain elevated soil organic matter stocks and replenish nutrient losses (Morón & Sawchik, 2003). However, crop rotation was abandoned over the last 15 years due to soybean expansion (García-Préchac et al., 2004). Nevertheless, more recently, new regulations for soil conservation have been successfully established in some countries of the region (e.g. Uruguay), with elevated adoption by farmers. Several parts of the region are experiencing decreases in light interception, and potentially their net primary production, with cascading effects on trophic networks. For example, large and consistent negative trends in net primary production have been observed in some parts of Uruguay and Argentina, associated with land use and climate change (Paruelo et al., 2004).

Southern temperate grasslands have been strongly invaded by plants and animals. The grass, *Eragrostis plana* was accidentally introduced into southern Brazil from Africa in the late 1950s (Guido & Pillar, 2017). Later planted as a potentially promising forage grass, it has now invaded over 1 million ha of grasslands (Medeiros et al., 2014). *Eragrostis plana* turned out to have low digestibility for cattle and causes economic losses by outcompeting more palatable native grasses. This is a very good example of how things can go wrong. Meanwhile, *Braquiaria* grasses (see also *Urochloa* spp.) are becoming adjusted to the local climate and could become a serious and widespread invasion problem in the future. The same grasses affect Uruguayan grasslands (Aber & Ferrari, 2010), so, without action, these invasions can be expected to expand in the coming years, encroaching on natural grassland areas. Exotic trees (e.g. *Gleditsia*, *Thriacanthos pines*) are also invading large areas of the region, altering grasslands physiognomy and displacing the local flora and fauna. Woody invasive species such as brooms (*Spartium junceum*, *Genista monspessulana* and *Ulex europaeus*), spiny rosaceous shrubs (*Rosa* spp. and *Rubus* spp.) and pines (*Pinus halepensis*, *P. radiata*) fit particularly well in a highly altered landscape matrix. Net forest cover in temperate grasslands increased from 2000 to 2012 (Figure 3.19), but this increase is attributed mainly to exotic tree plantations (Salazar et al., 2015).

Invasive vertebrates include wild boar (*Sus scrofa*). This species threatens key conservation habitats, affects agriculture and acts as a reservoir for diseases affecting pig farming, chital (*Axis axis*), and feral horses. European carp (*Cyprinus carpio*) has colonized most freshwater habitats, while common starlings (*Sturnus vulgaris*) and the red-bellied tree squirrel (*Callosciurus erythraeus*) are currently undergoing range expansion. Pet trade, forestry and aquaculture are emerging as new vectors of species introduction and expansion (see also Chapter 4). Other invasive animals include European pigeons, deer, and bullfrogs.

3.4.1.8 Drylands and deserts

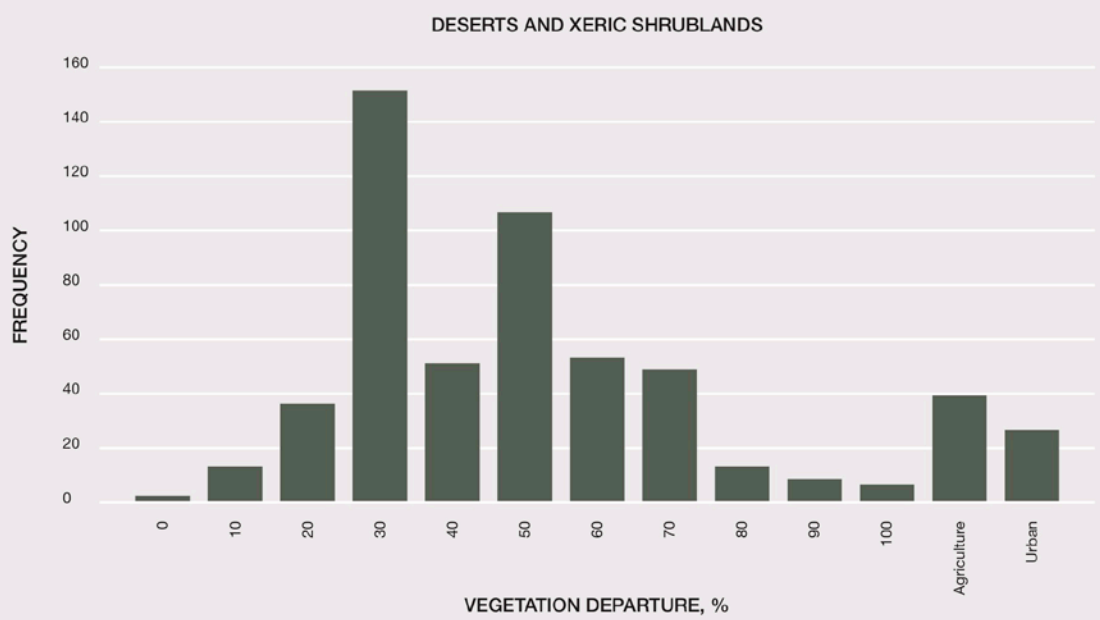
North American and Mesoamerican subregions

Drylands are ranked as one of the most important biomes for the biodiversity of species and endemics both globally and in the Americas (Goudie & Seely, 2011; Le Saout et al., 2013; Millennium Ecosystem Assessment, 2005). Much of the rich biodiversity and endemism (Table 3.4) found in these regions in the Americas and elsewhere is likely due to the high climate variability, which can drive speciation. High levels of endemism occur both at the species (Table 3.4) and generic levels. For example, 44% of seed plant genera in Mexican drylands under a broad definition are endemic (Challenger & Soberón, 2008). Animal biodiversity in North American can closely rival that found in tropical regions: Arizona alone contains 203 snake species (Southwestern Center for Herpetological Research, n.d.), almost two-thirds of the number found in the entire Amazon Basin. Unfortunately, many of these species have small home ranges, placing them at a high risk of extinction (Pimm et al., 1988). Biodiversity of lichens and mosses in dryland biological soil crusts, critical to soil stability and fertility, often exceeds vascular plants (Belnap et al., 2016).

Current habitat fragmentation, number of globally threatened animal species, and altered fire cycles in these drylands are rated moderate to very high (Hoekstra et al., 2010). In Mexican drylands, fragmentation is greatest in coastal deserts (Arriaga, 2009). One fragmentation index indicates that the largest mean parcel size of intact habitat in North America is only about 4% of the total extent of the dryland ecoregion (Figure 3.22). Nearly all drylands in North America and Mexico have been grazed by livestock relatively heavily at some point since European settlement; it is thus difficult to know how current ecosystems differ from those present before then. Estimates of departure of current vegetation conditions in the dryland biome relative to undisturbed dryland conditions based on the vegetation departure index are high in many areas of the biome, frequently more extreme than in agricultural or urban environments (Figure 3.22).

Figure 3 22 **Percentage of departure between current vegetation conditions and reference vegetation conditions of dryland desert and xeric shrublands (aridity index < 0.05 extracted from 30 arc second (~1km^2) resolution) and based on the VDEP index of the USA Forest Service and USA Geological Survey.**

Higher values indicate a greater departure from potential, or undisturbed vegetation. Agricultural and urban areas are grouped on the right for comparison. Source: Original data from The Nature Conservancy (2009) and The Nature Conservancy Terrestrial Ecosystems.



Box 3.7. . The Cuatro Ciénegas Basin in Coahuila, Mexico.

This ultra-low nutrient oasis in the Chihuahuan desert is extremely diverse, hosting at least 99 micro-endemic species and an equally wide array of microbial mats and stromatolites with ancestral marine lineages (CONABIO database, n.d.). The water's extremely low phosphorus content is characteristic of ancient ocean chemistry, earning it the nickname "Precambrian Park" (Redfield, 1934; Souza et al., 2012)**.** It exceeds diversity of other aquatic pools within desert systems globally by several orders of magnitude in the case of microbes and manyfold for other groups, such as spiders. Viral diversity is higher here than any other site in the world, reflecting the diversity of their bacterial and eukaryotic prey. The level of macrofauna endemism is equal to that of the Galápagos and is higher than anywhere in North or Mesoamerica (Stein et al., 2000)**.** Many species are new to science and still in the process of being described. The unusual geological history of this area explains its biodiversity: a large portion of the ancient Tethys Sea became entrapped by the regional uplift of the Sierra Madre Oriental and Occidental, isolating ancient seawater communities and leaving them to evolve independently (Ferrusquía-Villafranca et al., 2005; Souza et al., 2006, 2008, 2012)**.** Due to intensive agriculture, 90% of Churince, the most widely studied part of the basin, has disappeared since 2006, with most of the loss occurring in 2017; the remaining 10% is unique since most of the species are microendemic to the basin and their unique site. The whole Cuatro Ciénegas Basin is threatened by the intensified use of the deep aquifer for agriculture, causing water to be drained at a very rapid pace.

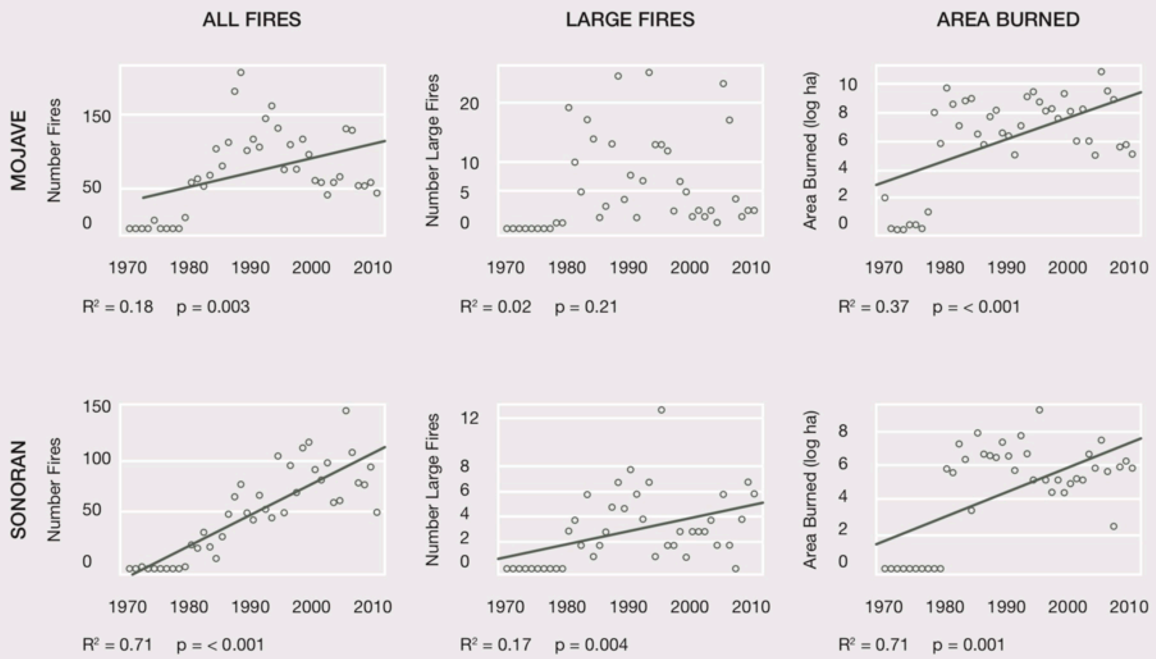
Dryland regions contain significant numbers of species that occupy habitats that have always had a very restricted range and thus are at high risk to disturbance. Reptile declines are associated with habitat loss. Individual desert tortoises occasionally move long distances between populations (Edwards et al., 2004), but movement is increasingly difficult for tortoises due to habitat fragmentation. The main cause of a decline in the bunchgrass lizard (*Sceloporus scalaris*) in the Chiricahua Mountains in southeastern Arizona, USA, has been attributed to the loss of native bunchgrasses due to cattle grazing (Ballinger & Congdon, 1996). This lizard requires bunchgrasses for cover and protection from predators and harsh winter conditions.

Recent trends. Habitat loss between 2000 and 2009 is estimated at 15–60% in North America (Challenger & Dirzo, 2009; Hoekstra et al., 2010). Biodiversity, soil health, and most associated ecosystem functions have declined over the past 50 years across these regions (e.g. http://www.biodiversitymapping.org; Goudie & Seely, 2011; Kéfi et al., 2007; Sarukhán et al., 2015). Biodiversity loss can be severe, as in the case of the highly specialized dune sagebrush lizard (*Sceloporus arenicolus*) of sandy depressions of dunes semi-stabilized by Shinnery oak (*Quercus havardii*) (Ryberg et al., 2014). These dunes are currently experiencing a large amount of energy exploration and development, resulting in their mobilization and thus severe loss of lizards and their habitat. A 13-year study of the twin-spotted rattlesnake (*Crotalus pricei*) found that the age class structure has been skewed toward younger snakes, probably due to illegal collection of snakes for the pet trade (Prival & Schroff, 2012). Unique ecosystems like the Cuatro Ciénegas Basin in Coahuila, Mexico (Box 3.7) have experienced recent losses of microbial biodiversity found nowhere else on Earth.

Loss of sagebrush habitat in the western USA has also impacted biodiversity, including the sage-grouse. This bird was once widespread and common, inhabiting, at the time of European settlement, what was a relatively uninterrupted vast (~46,521 km^2) sea of sagebrush (*Artemisia tridentata tridentata*) (http://sagemap.wr.usgs.gov). Due to agricultural cropping, fire, grazing, and energy extraction, this bird now occupies about 1/10 of its original range (~4,787 km^2) and is believed to be in peril of extinction. Rehabilitation of the sagebrush habitat has proven very difficult especially with the invasion of exotic annual Mediterranean grass *Bromus* (Germino et al., 2016) which accelerates fire cycles, leading to further loss of sagebrush on a large scale (Germino et al., 2016).

Figure 3 23 **Recent trends in fire frequency and area burned in the Mojave (upper) and Sonoran (lower) deserts based on data for southern California.**

Left: all fires; middle: large fires. Source: Modified from Syphard *et al.* (2017).



Exotic plants have increased in North American drylands due to several causes, but especially increased fire and soil surface disturbances; this invasion negatively impacts plant and animal communities (Brooks, 2009). Fire frequency and area burned increased in the Californian portions of the Mojave and Sonoran deserts between 1970 and 2010 (Figure 3.23). Exotic grasses, which burn easier than other vegetation types, were an important explanatory variable for large fires in the Mojave, but the amount of native perennial vegetation was more important in the Sonora (Syphard et al., 2017).

In the Sonoran desert, biological soil crusts have shown a dramatic decline in cover over the past 50 years, as they are highly vulnerable to fire and the disturbance of soil surfaces (Belnap & Eldridge, 2003). The loss of native plants, animals, and biological soil crusts has led to increased soil erosion via wind and water erosion; decreased soil albedo over large regions; and had a strong negative impact on water, carbon, and nutrient cycles (Ahlström et al., 2015; Fields et al., 2009; Hoekstra et al., 2010; Painter et al., 2010; Neff et al., 2005).

South American subregion

Status. Notwithstanding increasing intensive agriculture and urban encroachment, large parts of the Atacama and Sechura deserts in western South America remain fairly intact (Luebert & Pliscoff, 2006). The western deserts, although in large part seemingly barren, are an area of unexpected richness, especially in plants and microorganisms. Plant species-richness and endemism are especially high in the narrow coastal loma vegetation band (Dillon et al., 2011; Rundel et al., 1991; Squeo et al., 1998) (Table 3.4). Cactaceae are important and highly threatened (Goettsch et al., 2015; Guerrero et al., 2011; Larridon et al., 2014; Ortega-Baes & Godínez-Alvarez, 2006). Saline lakes and barren areas of the Atacama contain fascinating assemblages of Archaea, bacteria, and cyanobacteria (Crits-Christoph et al., 2016; Fernandez et al., 2016; Lester et al., 2007; Navarro-González et al., 2003; Wierzchos et al., 2006). Western deserts are subject to flash floods, and thus vegetation integrity plays a critical role in containing water erosion. Western coastal desert loma vegetation in particular, is highly susceptible to invasion when disturbed (Aponte & Cano, 2013). While some areas of the western deserts are under threat, a growing appreciation of the rich so-called "flowering desert" in Chile as a tourist resource has greatly heightened public awareness of the value of biodiversity.

Caatinga vegetation in eastern Brazil, part of this biome, is also rich (Table 3.4). Caatinga is poorly known in comparison to Brazilian Cerrado and tropical rainforest. The caatinga woody matrix is estimated to comprise around 63% of the original cover (Beuchle et al., 2015, but see Schulz et al., 2017) and thus is better conserved than Cerrado. Ten mammals are strictly endemic to caatinga and 11 more are endemic to the caatinga and Cerrado (Gutiérrez & Marinho-Filho, 2017). While most alien plant species in western drylands were accidentally introduced, the Caatinga is home to many intentionally introduced tropical forage grass species (Almeida et al., 2015).

Recent trends. Urban encroachment into the western loma vegetation has affected a highly endemic, range-restricted rodent to the point of likely global extinction of (Mena et al., 2007) warning that other local endemics in loma could be at risk with coastal development. A recent wave of private coastal development in Chile has greatly reduced the habitat of a globally threatened plant species (García-Guzman et al., 2012) and other species are likely affected. The production and the illegal extraction of Cactaceae continues (Estevez et al., 2010; Larridon et al., 2015). Extensive vegetation dieback, accompanied by declining guanaco populations, has been reported repeatedly over the last 20 years in the arid-most part of the western coastal desert (Schulz et al., 2011 and references therein). This trend coincides with a tendency for reduced precipitation, extended drought periods and reduced coastal cloud, notwithstanding typical El Niño variation. On the transition to Mediterranean shrublands, continuous monitoring has revealed El Niño Southern Oscillation-related fluctuations in the abundances in small mammals (Armas et al., 2016; Meserve et al., 2011) and alien plant species, but with significant recovery of native plants in wetter years (Jiménez et al. 2011), indicating high resilience at least in less arid areas. A noticeable shift in small mammals followed the last major El Niño Southern Oscillation event in 2000-2002 with their numbers becoming less fluctuating. This appears to have been caused by a shift in rainfall periodicity from strong interannual fluctuations, to a more equitable pattern with more consistent annual rainfall. These trends may be indicative of ongoing climate change in the Chilean semiarid region (Armas et al., 2016).

Biome-scale studies agree that the Caatinga has seen recent large-scale vegetation turnover and cover changes. However, both increases and decreases in woodland and woody vegetation have been reported. While studies based on Moderate Resolution Imaging Spectroradiometer data tend to find a net gain of woody vegetation, those based on Landsat data find a net decrease (discussed in Schulz et al., 2017). The impacts on caatinga species and populations of this highly dynamic scenario, to which a fertilization effect of carbon dioxide might be relevant (Donohue et al., 2013), are largely unknown. In addition to many introduced forage grasses, a serious ongoing invasion in caatinga concerns *Prosopis juliflora* which forms monospecific stands that outcompete native woody species and now covers over one million hectares (Gonçalves et al., 2015). As in the western deserts, selective biomass removal for fuel continues in the Caatinga, even though many households now possess gas stoves (Cavalcanti et al., 2015; Ramos & Albuquerque, 2012).

3.4.1.9 Wetlands: peatlands, mires, bogs

North American subregion

Status. North America houses approximately 240 millions of hectares of wetlands comprising 12.6% of the total land area. Some of the largest North American wetland landscapes are the peatlands of the Hudson Bay Lowlands, the peatlands of the Mackenzie River Watershed (Vitt, 2016), the Prairie Pothole region of the glaciated midcontinent of Canada and the USA, covering 7.7 million ha, and The Everglades and Great Cypress Swamp, covering 1 million hectares located on the southern part of the Florida peninsula. The boreal peatlands of Canada (110 million hecares), store an estimated one-third of the world's global carbon and 10% of the world's soil nitrogen (Vitt, 2016). The cold anaerobic conditions of boreal peatlands favor the accretion of undecomposed mosses, sedges, and other plants, together, resulting in deep organic deposits of 2m or more. Canadian peatlands support exceptional bryophyte diversity, with a recorded 294 species of mosses and related species (about one-third of the world's moss species) (Junk et al., 2006). The Prairie Potholes and the Everglades have outstanding biodiversity (Table 3.4). The Prairie Potholes provide critical breeding and migratory habitat for North America's waterfowl. The Everglades serve as a wintering area for 249 migratory bird species, as well as 100 resident species and critical habitat for species of global conservation concern.

From the 1800s to the 1980s Canada sustained losses of about 20 million hectares of wetland habitat. The conterminous USA sustained wetland losses of 53% (117 million hectares) from the 1780s to 1980s; Alaska lost less than 1% (Dahl, 1990). Despite these losses and much regional variation, wetlands still cover 12% of North America (240 million ha) (Dahl, 1990, 2011; Federal Provincial and Territorial Governments of Canada, 2010).

Recent trends. Losses to drainage for agriculture over the past 40–60 years has been the most important cause of wetland loss; conversion for development has also been locally significant near urban centers. An estimated 350,000 ha of wetland habitat in Canada was lost over the past 40–60 years, to drainage for agriculture in the prairie pothole region (Government of Canada, 2009). Wetland losses in Greenland are presumed to have been negligible. Compared to historic rates of wetland conversion, loss rates in both the USA and Canada have likely been lower in recent years because federal policies create disincentives for filling and draining wetlands (i.e., US Clean Water Act of 1972), Canada's Federal Policy on Wetland Conservation of 1991 (Government of Canada, 1991). Unfortunately, policies that allow compensatory restoration to offset conversions have not been effective at preventing losses of forested wetlands, which are costly and difficult to restore (Dahl, 2011).

Many North American wetlands have undergone extensive eutrophication. The associated changes of eutrophication include changes in the composition of aquatic life and recreational uses, in the effectiveness of wetlands as effective filters that protect downstream and groundwater resources, and in accumulation rates of bulk sediments (Brenner et al., 2001).

Wetland alteration has favored the expansion of invasive species and displacement of native species. Some serious wetland invaders in North America include common reed (*Phragmites australis*) in freshwater and brackish wetlands, cordgrass (*Spartina alterniflora*) in West Coast salt marshes and hybrid cattail (*Typha* x *glauca*), reed canary grass (*Phalaris arundinacea*) and purple loosestrife (*Lythrum salicaria*) in freshwater marshes. These invaders diminish wetland services in many ways including lost critical habitat for endangered species (e.g. *Phragmites*, central Platte River) and reduced wetland bird nesting (*Typha*, Great Lakes). Although a lot of attention and much funding have been devoted to managing and controlling these species, their spread is generally irreversible. Proximity to urban areas, as in the Everglades, has been associated with the escape and establishment of a large number of ornamental plants and pet animal species, including 221 plants, 32 fish, 30 amphibians and reptiles, and 10 mammals in the Everglades (Brown et al., 2006).

Many wetlands in urban areas that have been modified by filling or dredging experience high pulses of stormwater from watersheds with diminished infiltration, and receive toxins from transportation (e.g. chloride from road de-icing salts) and industrial run-off (Brinson & Malvárez, 2002; Sanzo & Hecnar, 2006; Federal Provincial Territorial Governments of Canada, 2010). Simultaneously the combination of alterations from urban development and agriculture has caused radical changes to the water quality and water flow in places like the Everglades.

Mesoamerican subregion

Status. The Mesoamerican subregion possesses an outstandingly diverse contingent of large tropical wetland areas with abundant bird, fish and large mammals (Table 3.4), among them the Centla Swamplands Biosphere Reserve south of the Gulf of Mexico, the Los Guatusos wetland area on the southern coast of Lake Nicaragua, and Palo Verde National Park and the Northeast Caribbean Wetland (Tortuguero) in Costa Rica (Hernández, 1999), all together summing to 141,470 km^2. The Centla Swamplands, located at sea level, constitute one of the world´s largest swamp areas. They are the home of gallery vegetation, mangroves, aquatic plants, manatees, jaguars, crocodiles, turtles and many fishes and birds. The Guatusos Wildlife

Refuge, with many fish species, is inhabited by indigenous and mestizo peoples. Like wetlands in general, it is a very important area for migratory birds, in the dry season in Nicaragua. Palo Verde National Park includes deltas, estuaries, flood plains, swamp forests and seasonally flooded grasslands. Counts of more than 50,000 waterfowl have been made in the wetlands of Palo Verde National Park, including the endangered Jabiru stork (*Jabiru mycteria*) (Daniels & Cumming, 2008). Tortuguero is dominated by herbaceous swamps and wooded palm-dominated floodplains that run parallel to the coast. It is an important site for nesting green turtles and several threatened species. In general palm-dominated wetlands in Costa Rica and Nicaragua constitute 16-22% of all wetlands (Serrano-Sandi et al., 2013); this type of wetland tends to be relatively poor in birds (Beneyto et al., 2013) as well as reptiles and amphibians (Bonilla-Murillo et al., 2013). An estimated 35% of Mexican wetlands have been transformed or suffered some level of deterioration (Hernández, 1999).

Recent trends. Contamination from heavy metals has been reported in the Centla swamplands (Pérez-Cruz et al., 2013), while pesticides related to agriculture have been reported in the Palo Verde Wetlands (Mena-Torres et al., 2014). Tabasco, where the Centla wetlands are located, is an area where petroleum extraction is currently occurring and is a threat to the reserve. The effect of these contaminants on aquatic biodiversity, however, is still unclear as baseline studies are only beginning. At the same time, the probability of wetland conversion increases as areas of wetlands are found closer to already converted land (Daniels & Cumming, 2008). Nevertheless, Landsat maps of Normalized Difference Vegetation Index suggest that the Palo Verde wetland has witnessed an overall increase in vegetation greenness and cover since 1986, matching the abandonment of cattle ranching and the known degradation of the wetland by cattail invasion (Alonso et al., 2016). This study shows that large degraded tropical Mesoamerican wetlands have the capacity to recuperate when external pressures are removed. The Tortuguero wetland is threatened by subsistence, sports fishing, poaching, the illegal collection of turtle eggs (Hernández, 1999) and pesticides from banana plantations and packing plants (Castillo et al., 2000). All these changes impact on human well-being. For example, total shrimp catch in El Salvador and Panamanian wetlands has dropped by 50% in the past decade or so (Hernández, 1999).

South American subregion

Status. South American wetlands are hugely diverse, spread over the entire continent, and found from sea level to above 5000 m altitude. The three largest wetland areas (Amazon river basin, Pantanal, Magellanic peatlands) in accordance with their sizes (Keddy et al., 2009), comprise around 11% of South America. Other large wetlands are the Orinoco delta with large peatlands and the internal Venezuelan and Colombian deltas, which are pantanal-like areas. Total wetland extent is difficult to pin down, given lack of consensus over what constitutes a wetland and the fact that some wetlands tend to be overlooked. A case in point are the Veredas of Brazil, spread over the entire savanna biome and perhaps comprising as much 5% of that biome. Many wetland types are rich in bird species (Caziani et al., 2001; Derlindati et al., 2014; Mascitti & Bonaventura, 2002; Tellería et al., 2006) (Table 3.4) and have important aesthetic value. Amazonian flooded forested wetlands and the Pantanal are especially rich in plants, birds, fishes, reptiles, and amphibians (Table 3.4). Some wetlands are rich in planktonic assemblages (Küppers et al., 2016; Muñoz-Pedreros et al., 2015). In general, South American wetlands play a vital role in water regulation for surrounding forests and agricultural lands. For example, certain types of subantarctic peat bogs and mires, given the high water-holding capacity of *Sphagnum* species and of accumulated peat, discharge water slowly and have an important buffering effect on surrounding forest ecosystems (Iturraspe, 2010). Southern South American peatlands, including Amazonian and (and probably Orinocan peatlands), are important carbon sinks (Lähteenoja et al., 2009; Loisel & Yu, 2013). In contrast, tropical floodplain lake ecosystems with a large amount of organic matter are considered important sources of carbon from the water to the atmosphere (carbon dioxide evasion) (Raymond et al., 2013), although abundant macrophytes can counteract this effect locally (Peixoto et al., 2016), indicating an important biodiversity link. High Andean bogs in the arid puna are important for the grazing of native camelid and other domestic animals which sustains the livelihoods of high altitude peoples (Borgnia et al., 2008; Tirado et al., 2016) (see also Box 3.6).

Recent trends. Many South American wetlands have been severely degraded over the past 30–40 years. High Andean bogs and associated salt lakes in arid areas of the Andes are now used extensively as a water source for mining (Aitken et al., 2016), although this source is now being replaced by imported seawater from adjacent coastal areas in some cases. Roads built over these fragile ecosystems are an additional problem (Salvador et al., 2014). Water levels in the bog complex in the arid Andes are critical for bird species maintenance (Tellería et al., 2006). Water withdrawal also alters a key habitat for grazing animals. In the vast Pantanal, largely as a result of large-scale cattle ranching and cropping, between 1976 and 2008 loss of floodplain vegetation increased over 20-fold (Silva et al., 2011) with some 12% lost. Loss of pantanal has lead to negative consequences for large animal species (Keuroghlian et al., 2015). Nevertheless, absolute loss of pantanal is much lower than for Cerrado. In the lower Paraná delta in Argentina, between 1999 and 2013 one-third of the freshwater marshes (163,000 ha) were replaced by cattle pastures (70%) and forestry (18%) (Sica et al., 2016) over a period of no more than 14 years. As of the 1970s, intensive commercial fisheries developed across the Amazon and the lower Paraná delta. Overexploitation of frugivorous fish species has depressed the quantity, quality, and diversity of seeds dispersed by fishes which could lead to overall reduced plant diversity in these habitats (Correa et al., 2015). Long-term studies (1969–1987) in the extra-tropical Mar Chiquita in Argentina reveal that flamenco breeding is very susceptible to lake water levels, especially excessive flooding (Bucher et al., 2000), making this kind of wetland vulnerable to surrounding land use changes affecting upstream flow. While commercial peat extraction is still limited in Magellanic peat bogs, abandoned peatlands show a significant invasive plant species component (Domínguez et al., 2012). The integrity of southern peat bogs is further threatened by introduced beaver, which increased in number from 54,000 to 110,000 between 1999 and 2015 (Instituto de Asuntos Públicos-Centro de Análisis de Políticas Públicas, 2016).

3.4.1.10 Summary biodiversity data for terrestrial biomes and overall trends for terrestrial biomes and other units of analysis

Table 3.4.Illustrative biodiversity data for principal terrestrial biomes in subregions of the Americas. The first number in parentheses gives richness; % value is endemism level where available.

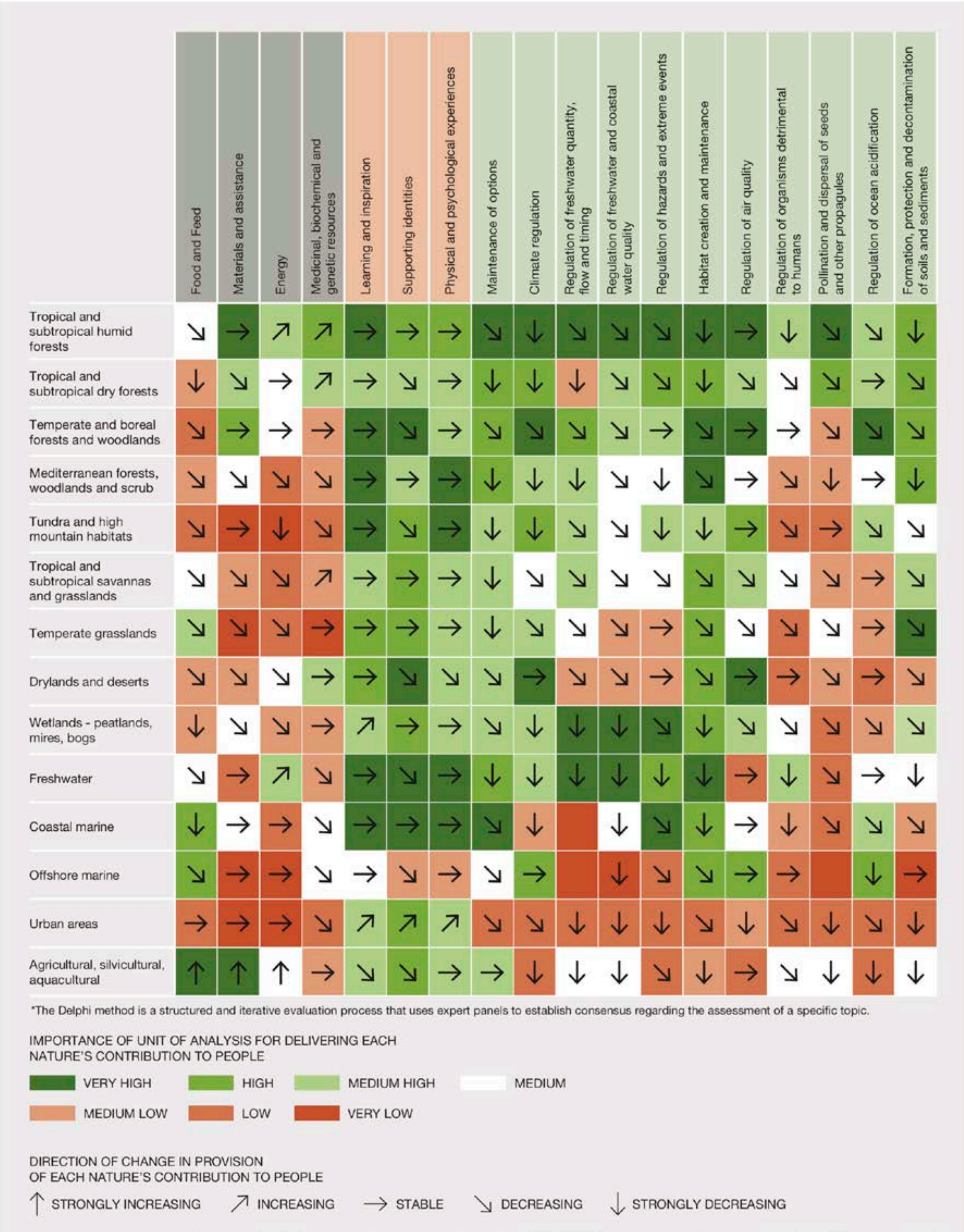
North American subregion
Temperate and boreal forests and woodlands
USA forests: plants (9,195), mammals (234), birds (452), reptiles (218), amphibians (201), freshwater fishes (60), invertebrates (739), trees (~1,000) (U.S. Forestry Service, 2015)**.**
Mediterranean forests, woodlands and scrub
California Floristic Province: plants (5,006; 37%) (Burge et al., 2016); California: mammals (201), birds (653; 1%), reptiles (101;15%), amphibians (70; 46%) (Zavaleta *et al.*, 2016), bees (1,600) (Frankie *et al.*, 2014)**.**
Tundra and high mountain habitats
North American tundra: vascular plants (1,486) (Elven et al., 2011), mammals (41), birds (152), amphibians (1), insects (1,567), spiders (200), springtails (174), mites (368), white worms (73) (Meltofte, 2013); Western North American alpine: plants (> 1,400) (Malanson *et al.*, 2015).
Temperate grasslands
Midwestern grasslands: plants (897) (Wilsey *et al.*, 2005).
Drylands and deserts
Mojave Desert of southern California: plants (5,000) (USDA n.d).
Wetlands, peatlands and mires
Canadian peatlands: mosses and related species (294) (Junk et al., 2006); Everglades: plants (1,033), birds (349, 249 migratory), fishes (432), reptiles (60), mammals (76), amphibians (38) (Brown et al., 2006), macroinvertebrates: 290–400 (Trexler & Loftus, 2016).
Mesoamerican subregion
Tropical and subtropical moist forests
Mexico lowland tropical broadleaf forest: seed plants (~5,000) (Challenger & Soberón, 2008); Mexican montane mesophyll forest: seed plants (~3000) (Challenger & Soberón, 2008); Mexican coniferous forest: pines (54), oaks (160) (CONABIO, 2014); Eastern Panama broadleaf forest: mammals (~165) (Voss & Emmons, 1996); Southern Mexico: mammals (~125) (Voss & Emmons, 1996).
Tropical and subtropical dry forests
Costa Rica: plants (~4,500), vertebrates (~1,100), arthropods (~150,000), fungi (~20,000) (Janzen, 1987; Janzen & Hallwachs, 2016); Mexico: seed plants (~6,000; 40%) (Challenger & Soberón, 2008), trees (1,072) (Banda-R *et al.*, 2016); Central America (northern South America included): trees (808) (Banda-R *et al.*, 2016).
Drylands and deserts
Mexico: seed plants (~6,000) (Challenger and Soberón 2008), endemic plants (3,600) (Arredondo Moreno & Huber-Sannwald, 2011), cacti (550 spp.; 78%) (Goettsch & Hernández, 2006).
Wetlands – peatlands, mires, bogs
Mexico, Centla Swampland: birds (213) (Santiago-Alarcon *et al*., 2011), fishes (44) (Macossay-Cortez *et al*., 2011); Nicaragua, Guatusos Wildlife Refuge: mammals (32), birds (>300), reptiles (10) (Hernández, 1999).
Caribbean subregion
Tropical and subtropical moist forests
Caribbean Islands (all terrestrial ecosystems): plants (11,000; 72%), mammals (69; 74%), birds (564; 26%), reptiles (520; 95%), amphibians (189; 100%), freshwater fishes (167; 39%) (Wege *et al.*, 2010).

Tropical and subtropical dry forests
Woody plants (611) (Banda-R *et al.*, 2016).
South American subregion
Tropical and subtropical moist forests
Amazonia: plant species (14,003), trees (6,727) (Cardoso *et al*., 2017), trees (11,676) (ter Steege *et al*., 2016)**,** birds (1,300) (Marini & Garcia, 2005), reptiles (378), amphibians (428), fishes (>3,000) (Charity *et al*., 2016); Amazonian lowland forest: mammals (434) (Mares, 1992); Atlantic Coastal forest: plants (~20,000), mammals (263), reptiles (306), amphibians (475) (Mittermeier *et al*., 2005), birds (1,020) (Marini & Garcia, 2005); Andean Montane forest: trees (3,750) (Tejedor Garavito *et al.*, 2015), birds (many of a total of 1,160 species in all neotropical wet montane forests) (Stotz *et al*., 1996), mammals (332) (Mares, 1992); Las Yungas, Bolivia: plants (6,073) (Jørgensen *et al*., 2015).
Tropical and subtropical dry forests
Northern South America and Central America: tree species (808) (Banda-R *et al*., 2016); Northern interandean Valleys: trees (418) (Banda-R *et al.*, 2016); Colombian dry forest: plants (2,569), birds (230), mammals (60) (Gómez *et al.*, 2016).
Temperate and boreal forests and woodlands
Temperate rainforests: plants (443–500) (Arroyo *et al.*, 1996; Villagrán & Hinojosa, 1997), mammals (58), birds (60) (Armesto *et al*., 1996); Magellanic rainforest–tundra zone: bryophytes (450), liverworts (368) (Rozzi *et al.*, 2008); Tierra del Fuego and Patagonia: myxomycetes (67) (Wrigley *et al*., 2010).
Mediterranean forests, woodlands and scrub
Central Chile: vascular plants (2,900; 30%) (Arroyo *et al*., 2002), mammals (37), birds (200), reptiles (38), amphibians (12) (Simonetti, 1999), bees (~300) (Montalva & Ruz, 2010).
Tundra and high mountain habitats
Whole biome: plants (6,700) (Arroyo & Cavieres, 2013); Páramo: vascular plants (3,600) (Sklenář *et al*., 2005), non-vascular plants (1,300) (Luteyn, 1999); Puna freshwater and salt lakes: fishes (60) (Vila *et al.*, 2007).
Tropical and subtropical savannas and grasslands
Brazilian Cerrado: plants (13,137), birds (837) (Overbeck *et al.*, 2015), mammals (251) (Paglia *et al*., 2012), trees (2,916) ("Tree flora of the Neotropical Region," n.d.)**.**
Temperate grasslands
Río de la Plata grasslands: grass species (550) (Bilenca & Miñarro, 2004).
Drylands and deserts
Chilean winter rainfall deserts (broadly): plants (1,893) (Arroyo & Cavieres, 1997); Pacific Coastal Lomas: plants (1,200) (Dillon *et al*., 2011); Caatinga: plants (2,400-4,230) (Moro *et al*., 2014), fishes (185), lizards (44), amphibians (8), snakes (47), turtles (4), crocodilians (3), amphibians (49) (WWF, 2017b), birds (519) (Silva *et al*., 2003), mammals (148) (Oliveira, 2003).
Wetlands: peatlands, mires, bogs
Amazonian wetlands: plants (>1,390), endemic trees (68) (Junk *et al*., 2014); Brazilian Pantanal: plants (1,863), aquatic and terrestrial mammals (170), bats (46-floodplain), birds (655 floodplain and uplands), herpetofauna (135 Plains), fishes (263) (Alho, 2011; Alho, *et al.*, 2011a; Alho *et al.*, 2011b; Pott *et al*., 2011)**.**

Figure 3 24 **Historical and recent habitat change and recent species trends for terrestrial biomes and other units of analysis considered in the assessment for the four subregions of the Americas. Source: own representation.**

		HISTORICAL TRENDS	RECENT TRENDS (40 YRS)				
	Units of analysis	Habitat amount	Habitat amount	Habitat degradation	Native species diversity	Threatened species	Alien & Invasive species
NORTH AMERICA	Temperate and boreal forests and woodlands	↔··	↗····	↗····	↘···	↔····	↗···
NORTH AMERICA	Mediterranean forests, woodlands and scrub	↓····	↘····	↗····	↘····	↗····	↗····
NORTH AMERICA	Tundra and high mountain habitats	↔··	↔··	↗····	↔····	↔····	↗··
NORTH AMERICA	Temperate grasslands	↓····	↘····	↗····	↘····	↗····	↗····
NORTH AMERICA	Drylands and deserts	↓····	↘····	↗····	↘····	↗····	↗····
NORTH AMERICA	Wetlands - peatlands, mires, bogs	↘····	↘····	↗····	↘·	↗·	↗····
NORTH AMERICA	Inland surface waters and water bodies / freshwater	↘····	↘·	↗····	↘···	↗···	↗····
NORTH AMERICA	Coastal habitats and nearshore marine	↓····	↘····	↗····	↘···	↗··	↗····
NORTH AMERICA	Cryosphere / Sea Ice	↔·	↘··	↗·	↔··	↔··	↗··
MESOAMERICA	Tropical and subtropical moist forests	↓····	↘····	↗··	↔··	↗··	↔··
MESOAMERICA	Tropical and subtropical dry forests	↓····	↘····	↗···	↘··	↗··	↗··
MESOAMERICA	Drylands and deserts	↘··	↘··	↗·	↘·	↗··	↗··
MESOAMERICA	Wetlands - peatlands, mires, bogs	↘··	↘···	↗·	↘·	↗··	↗··
MESOAMERICA	Inland surface waters and water bodies/freshwater	↘··	↘···	↗·	↘··	↗··	↗·
MESOAMERICA	Coastal habitats and nearshore marine	↘··	↘···	↗··	↘··	↗·	↔·
MESOAMERICA	Marine/deepwater/ offshore systems	↘··	↔·	↗·	↔·	↗·	↔·
CARIBBEAN	Tropical and subtropical moist forests	↘····	↗···	↔···	↔···	↔··	↗··
CARIBBEAN	Tropical and subtropical dry forests	↓····	↗··	↗···	↔··	↗··	↗··
CARIBBEAN	Inland surface waters and water bodies/freshwater	↔·	↘··	↗··	↘··	↗·	↗·
CARIBBEAN	Coastal habitats and nearshore marine	↔··	↓····	↑····	↔···	↗··	↗···
CARIBBEAN	Marine/deepwater/ offshore systems	↔··	↔···	↗···	↔···	↗··	↔·
SOUTH AMERICA	Tropical and subtropical moist forests	↘····	↘···	↗···	↘···	↗···	↗···
SOUTH AMERICA	Tropical and subtropical dry forests	↓····	↘···	↗···	↘··	↗···	↗···
SOUTH AMERICA	Temperate and boreal forests and woodlands	↘···	↘···	↗···	↘··	↗··	↗···
SOUTH AMERICA	Mediterranean forests, woodlands and scrub	↓····	↘····	↗···	↘···	↗····	↗····
SOUTH AMERICA	Tundra and high mountain habitats	↘···1 ↔···2	↘··	↗···	↔··	↔··	↔···
SOUTH AMERICA	Tropical and subtropical savannas and grasslands	↘····	↘····	↗··	↘··	↗···	↗···
SOUTH AMERICA	Temperate grasslands	↓····3 ↔····4	↔····	↗····	↔····	↔····	↗····
SOUTH AMERICA	Drylands and deserts	↔···5 ↘6····	↔··	↗···	↔··	↗···	↗···
SOUTH AMERICA	Wetlands - peatlands, mires, bogs	↔··	↘···	↗···	↘·	↘·	↗··
SOUTH AMERICA	Inland surface waters and water bodies / freshwater	↘··	↘··	↗··	↔··	↗··	↗··

1: Páramo and puna; 2: Other areas of biome; 3: Río La Plata Grasslands; 4: Other areas of biome; 5: Western deserts; 6: Caatinga.



3.4.2 Marine and ocean systems

Status. Considerable numbers of marine mammals are threatened in each of the four subregions (Table 3.5). Extinctions in the Americas include Steller's sea cow (*Hydrodamalis gigas*) native to the Bering Sea; the Caribbean monk seal (*Neomonachus tropicalis*) native to the Caribbean Sea, Gulf of Mexico and West Atlantic Ocean; and the sea mink (*Neovison macrodon*), native to coastal eastern North America (Committee on Taxonomy, 2016).

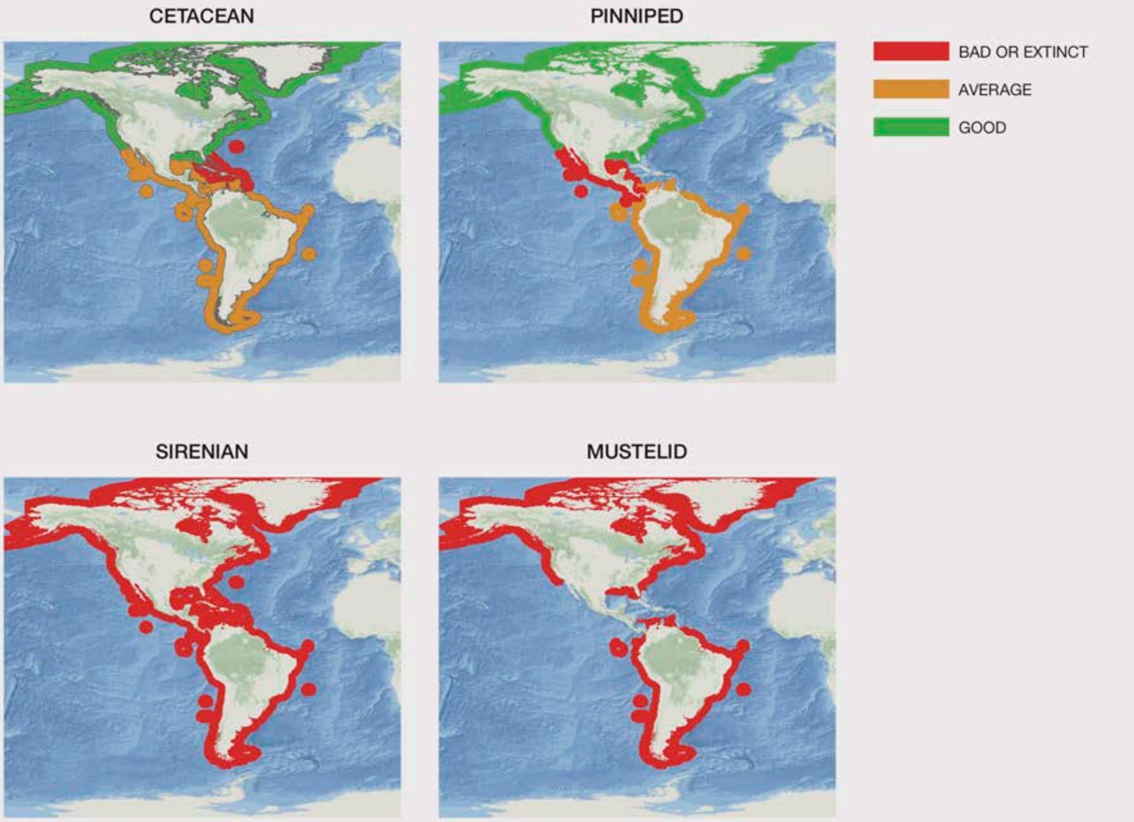
Table 3.5. **The number of marine mammal species found across the Americas, grouped by current IUCN Red List status and by subregion. DD = data deficient, LC = least concern, NT = near threatened, V = vulnerable, E = endangered, CE = critically endangered. Note three extinct species captured in these counts: Caribbean monk seal (*Neomonachus tropicalis*), Steller's sea cow (*Hydrodamalis gigas*), and the sea mink (*Neovison macrodon*). From IUCN Red List IUCN (2017).**

IUCN Status	Americas total	North America	Caribbean	Mesoamerica	South America
Data deficient (DD)	43	21	13	18	36
Least concern (LC)	35	27	10	10	18
Near threatened (NT)	4	3	0	0	1
Vulnerable (V)	7	6	2	2	3
Endangered (E)	10	7	3	3	6
Critically endangered (CE)	1	0	0	1	0
Extinct	3	2	1	0	0

Across subregions, trends in mammal populations are mixed (IUCN, 2017) (Figure 3.26). For example, although some sea otter populations are stabilizing or increasing, abundances remain below carrying capacity (Doroff & Burdin, 2015). Both extant manatee species are considered vulnerable with decreasing populations. Because suitable sea ice habitat in the Arctic is degrading and/or disappearing rapidly with climate change, the polar bear is considered vulnerable; however the trends across the 11 populations of polar bear are mixed (4 increasing, 2 stable, 5 decreasing), and trends across eight other subpopulations are unknown (IUCN & SSC PBSG, 2017; Wiig et al., 2015). Very little is known about population trends of most beaked whales and most dolphin species. Half of the turtle subpopulations that forage and/or nest in the Americas are endangered or critically endangered (IUCN, 2017).

Figure 3 26 **Population status for each type of marine mammal categorized according to species population trends.**

"Bad or extinct" (red) indicates most or all species are declining; "Average" (orange) indicates some species are in decline, some are stable, some are increasing and some are unknown; "Good" (green) indicates most species are increasing, stable or unknown. Not shown are extinct species, or the polar bear (Ursus maritimus) only found in the North America region (IUCN Red List status is "vulnerable" and the population trend is unknown). Source: Produced from status and trends species-level information in the IUCN Red List (2017).



Recent trends. In North America, protection under the US Endangered Species Act, the US Marine Mammal Protection Act, and the International Whaling Commission has led to increasing populations of some marine mammals (e.g. gray whales) and sea turtle species in USA waters, but habitat destruction and human activities continue to place other species in jeopardy. For example, the western North Atlantic right whale and Hawaiian monk seal continue to decline (Hourigan, 1999). Similarly, marine mammal populations in Canada are increasing, including grey seals in the Scotian Shelf and Gulf of St. Lawrence, harp seals in the Gulf of Maine and Scotian Shelf, western Arctic bowhead whales in the Beaufort Sea, Stellar sea lions, sea otters, and the Pacific harbour seal. Resident killer whale (*Orcinus orca*) populations off the coast of Vancouver Island have shown variable patterns since 2001, with the threatened northern population showing slight signs of recovery but the endangered southern population showing little recovery and listed as "endangered" in the USA "at risk" in Canada (Fisheries and Oceans Canada, 2017).

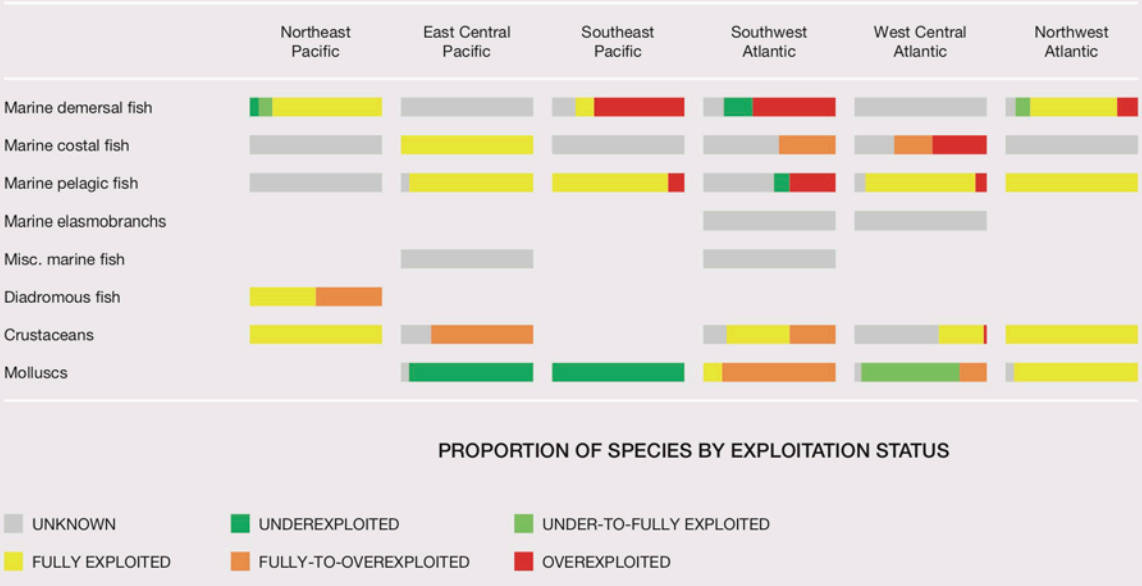
Across the Americas, despite bycatch reduction efforts, particularly in North America, some large whales are still endangered (e.g. North Atlantic blue whale, *Balaenoptera musculus*) as are cetacean populations with low abundances (Read, 2008). Some populations are small in number from previous anthropogenic impacts, such as false killer whales (*Pseudorca crassidens*). Sirenians (e.g. manatee, *Trichechus manatus*) and large whales are particularly vulnerable to fisheries bycatch and other types of removals because of inherent life history traits (e.g. slow maturation) that limit their potential population growth rate (Eberhardt & O'Shea, 1995). In Mesoamerica, the Caribbean and South America there is generally a lack of consistent, robust fisheries bycatch reduction management and/or enforcement, and fisheries bycatch remains a primary anthropogenic threat (Hucke-Gaete & Schlatte, 2004; Read, 2008). The vaquita, a small porpoise endemic to a small range in the northern Gulf of California, is an example of a small, critically endangered population subject to high bycatch rates; as such, this species is predicted to go extinct by 2022 or sooner (Taylor et al., 2017).

Around 338 marine time series of change in the Americas have been collected. These studies are distributed inequitably and are geographically sparse, with only eight in South America (Dornelas et al., 2014; Dunic, 2016; Elahi et al., 2015). Further, most time series are less than 10 years, precluding a comprehensive picture of how marine biodiversity has changed in the Americas over the past 40-50 years. The only ecoregion (Spalding et al., 2008) of the Americas with a sufficient sample size – the Southern California Bight, with 154 different available time series – shows a trend toward a general increase in local species diversity. Multiple studies show many marine species moving poleward, on average often in relation to shifts in ocean temperature (Cheung et al., 2013; Pinsky et al., 2013; Poloczanska et al., 2013; Sorte et al., 2010). Given high coastal diversity at low latitudes, this suggests that diversity in the future should increase outside of the tropics. Areas with extremely high cumulative human impacts (Halpern et al., 2008) have tended to show losses in diversity over time (Elahi et al., 2015).

Fisheries species (fish and invertebrates). Commercial fisheries occur in all oceans surrounding the Americas. Nearly all marine animal phyla as well as seaweeds are harvested in commercial fisheries, but fished taxa and recorded landings data are heavily biased towards fishes—both ray-finned fishes and cartilaginous fishes, and invertebrate animals, especially crustaceans such as lobsters, crabs, and shrimps; molluscs such as clams, abalones and squids; and echinoderms such as sea cucumber and sea urchins. Major fishing countries in terms of total landings include Peru, the USA, Chile, Mexico, Canada, Argentina, and Brazil.

While extinction risk is generally very low for marine fishes, recovery of marine populations may take several decades to recover even when fishing intensity is relaxed (Neubauer et al., 2013). In the Northeast Pacific and Northwest Atlantic, important fished species in most taxonomic groups are fully exploited (Figure 3.27), i.e., they are fished at levels near maximum sustainable yield, with annual, sustainable catches near optimal levels (Costello et al., 2016; Worm et al., 2009). These evaluations are largely based on quantitative stock assessments, which yield relatively low uncertainty in estimates of exploitation status. Stock assessments are, however, typically conducted for species with large volumes of fishery landings or species with high ex-vessel prices so they are not representative of all marine taxa.

Figure 3 27 **The proportion of fished species impacted by exploitation in different ocean regions adjacent to the Americas as determined by the FAO for individual species or species groups, which are subsequently aggregated into broad taxonomic groups. Source: Based on data provided by FAO (2012).**



Fewer species from northern latitudes are considered to be either over-exploited or under-exploited (Figure 3.27). A small proportion of Atlantic demersal fish species is overfished while a small proportion is underfished on both coasts of North America (Figure 3.27). Moving towards the tropics, in the east-central Pacific most coastal and pelagic fish species are fully exploited, most crustaceans are fully-to-overexploited, and molluscs are underexploited. With the exception of pelagic fish species, however, many of these categorizations are highly uncertain (FAO, 2016). In the west-central Atlantic, a higher proportion of coastal fish are overexploited, fewer crustaceans are overexploited, and more molluscs appear to be overexploited compared to fisheries in the Pacific Ocean although the latter estimates are highly uncertain. Moving further south, we find that most important demersal fish species are overexploited on both coasts. Most pelagic fish tend to be exploited in the southeastern Pacific while all are exploited in the southwestern Atlantic. On the eastern coast of North America, many of the offshore fisheries exceed target levels and are not considered sustainable, especially those of elasmobranchs (Brick Peres et al., 2012; Ministério do Meio Ambiente, 2006). Molluscs are underfished in the southeastern Pacific (though estimates are highly uncertain), while crustaceans and molluscs are fully exploited or overexploited in the southwestern Atlantic. The exploitation status of many species is unknown across several taxonomic groups, in particular, elasmobranchs (sharks, skates and rays) and coastal fishes.

Despite the collapse of certain fisheries, considerable efforts have been undertaken to manage fisheries in North America. Compilations of quantitative stock assessments (Costello et al., 2016; Worm et al., 2009) show that over-fished populations usually recover after fishing pressure is reduced. In the USA, management actions have resulted in a number of successes, including Alaska groundfish, king and Spanish mackerel, striped bass, and ocean quahogs (Hourigan, 1999). Only a small percentage of USA fisheries are now considered overfished. However, fisheries impact nontarget species through bycatch and seafloor damage by trawls (Watling & Norse, 1998). In Canada, an expert panel (Hutchings et al., 2012) concluded that marine fishes in Canada declined by an average of 52% from 1970 to the mid 1990s and then remained stable; most stocks, including some populations of groundfish, such as Atlantic and Pacific cod, lingcod and rockfish species, pelagic fish such as herring and capelin, and anadromous fish such as coho, Chinook salmon, Atlantic salmon and Arctic char remain well below target levels.

Three of the seven global threat hotspots for neritic and epipelagic oceanic sharks in coastal waters are in the Americas (Gulf of California, southeast USA continental shelf, Patagonian shelf) (Dulvy et al., 2014). Brazil, Mexico, Argentina and the USA are in the top 10 countries reporting the highest landings of chondrichthyans between 2003 and 2011 (Davidson et al., 2016). Currently, despite decades of population declines for many chondrichthyans, only 18 sharks and rays have been listed by CITES (Convention on International Trade in Endangered Species). Stock assessments for a number of chondrichthyans in the Americas report declines of 20–80% from unfished conditions for multiple species (Highly Migratory Species Management Division, 2006). In the eastern central Pacific, chondrichthyan landings steadily increased throughout the latter half of the 20th century, peaking to ~50,000 tonnes in 2000, and declining to <40,000 tons in recent years (FAO, 2011). Mexican catches (which represent >60% of regional chondrichthyan landings) have continued to increase, and current fishing practices targeting elasmobranch aggregations on breeding and pupping grounds are posing increased threats to many species (Kyne et al., 2012). Historic and current fishery landings data are limited, and the population status of most shark species throughout the region is poorly understood (Kyne et al., 2012). However, fishery surveys suggest that two species of sawfish – the largetooth sawfish and the smalltooth sawfish – may have experienced local extinctions in Belize and possibly Guatemala (Kyne et al., 2012).

Canada has become one the world's third-largest exporters of shark meat, and the USA has experienced the second greatest increase in chondrichthyan landings since 2003 (Davidson et al., 2016). The FAO recently identified Brazil as having one of the largest and most rapidly expanding shark product consumer markets in the world (Barreto et al., 2016). Some 32% of all Brazilian chondrichthyans are endangered and two species of shark are considered regionally extinct, according to IUCN Red List criteria (Reis et al., 2016). The southeastern coast of South America is also considered a hotspot of deepwater threatened chondrichthyans (Dulvy et al., 2014). Targeted shark fisheries have also expanded in Mexico and Venezuela (Tavares & Lopez, 2009).

Fisheries management plans are now in place for many elasmobranchs in the northwestern Atlantic, but lacking in most other areas. Recently, Chile, Colombia, Ecuador, and Peru have developed a regional action plan for protecting and managing chondrichthyans (Davidson et al., 2016). Recently completed stock assessments for two shark species in the northeastern Pacific also revealed that all populations are either not overfished or are recovering from historical overfishing (Kleiber et al., 2009; Tribuzio et al., 2015; Young et al., 2016).

3.4.2.1 Coastal habitats/Coastal and near shore marine/inshore ecosystems

Coastal marine habitats provide many ecosystem services, including food, protection against coastal erosion, recycling of pollutants, climate regulation and recreation.

Salt marshes

Status. Salt marshes are intertidal ecosystems that are regularly flooded with salt or brackish water and dominated by salt-tolerant plants. They remove sediment, nutrients and other contaminants from runoff and riverine discharge (Gedan et al., 2009), protecting estuarine biota. They also protect coastal communities from storm waves (Costanza et al., 2008) and are nursery areas for many commercial fish species. Many migratory shorebirds and ducks use salt marshes as stopovers during migration, and some birds winter in marshes. Wading birds, such as egrets and herons, feed in salt marshes during the summer. After European settlement, North American salt marshes were filled for urban or agricultural development or garbage dumps. Using historical maps, Bromberg & Bertness (2005) estimated the average loss in New England at 37%. Rhode Island has lost the most, 53%, since 1832. Salt marshes are estimated to have occupied 200,000 to 400,000 ha in pre-settlement Louisiana, with an estimated 50–75 % remaining (Smith, 1993) as of two decades ago. San Francisco Bay has seen a 79% reduction in its salt marshes. Salt marshes in South America have been far less drained (6%) than in North America (~50%) (Zedler & Kercher, 2005). However, these marshes are threatened by agriculture, construction of flood control measures and hydroelectric power, pollution, and large-scale fish and shrimp aquaculture. Some marshes on the Atlantic Coast of South America have extensive bare areas dominated by high densities of the crab *Chasmagnathus granulata* (up to 60 individuals /m2), which consumes the marsh grass *Spartina densiflora*. The bare areas, often comprising half of the habitat, are due to crab herbivory. It is suspected that the high densities of *Chasmagnathus* are at least in part due to the overfishing of predators (Bortolus et al., 2009). In South America, invasive *Spartina* species are found in coastal marshes (Orensanz et al., 2002).

Recent trends. In recent years, sea level rise has begun to impact many previously healthy marshes in the Americas (such as ponding, where water remains on the marsh surface during low tide and plants get waterlogged). The actual rate of sea level rise in the future will affect which marshes can persist. Other marshes are being restored, a very expensive procedure. There are some attempts to raise their elevations (Ford et al., 1999), yet given accelerating sea level rise, extensive areas will most likely continue to be lost. The invasive reed, *Phragmites australis*, which has reduced plant diversity in many brackish marshes in the eastern coast of the USA and is often removed in restoration projects, allows marshes to increase their elevation more rapidly (Rooth & Stevenson, 2000) and might better enable marshes to keep up with sea level rise. While 50% of the salt marsh area in New England had been lost by the mid-1970s, recent loss rates have been lower because of awareness of their value and restoration projects (Valiela, 2006). Long-enrichment of coastal salt marshes has reduced belowground organic matter, contributing to subsidence (Turner et al., 2009).

Mangroves

Status. In tropical and subtropical regions, intertidal mangroves perform similar ecological functions as salt marshes in temperate zones. The red mangrove, *Rhizophora* spp, lives at the water's edge with its aerial prop roots in the water, serving as the substrate for a community of attached invertebrates and shelter for fishes that swim among the roots. Caribbean mangroves are reported to host the world's richest mangrove-associated invertebrate fauna worldwide (Ellison & Farnsworth, 1996). Mangroves provide many NCP such as wood products, microclimate regulation, shoreline protection, nutrient cycling and carbon storage (Vo et al., 2012).

Recent trends. Recently, use of mangroves has increased leading to substantial loss (Valiela et al., 2001). Construction of shrimp and fishponds for aquaculture accounts for over 50% of the world's mangrove loss. In the Americas, losses average about 2.1% per year, with annual losses up to 3.6% per year. This is likely due to exploitation, deteriorating water quality, coastal development and climate change (Gilman et al., 2008; McKee et al., 2007; Polidoro et al., 2010). In the Caribbean, mangrove area has declined by about 1% annually over the last 30yrs, the second highest rate of loss globally (FAO, 2007). In recent years, mangroves have been spreading northward in Florida, expanding their range in response to warming (Cavanaugh et al., 2014). Since they are not likely to be harvested for wood or removed for aquaculture, this northward move may counterbalance some of the threats.

Submerged aquatic vegetation

Status. Seagrasses live submerged in salt or brackish water full-time and provide habitat for animals such as scallops, and, in tropical regions, juvenile coral reef fishes. USA populations crashed in the 1930s due to disease and slowly recovered over subsequent decades. Since the 1960s, much of the Submerged aquatic vegetation disappeared in North Atlantic estuaries, particularly in Chesapeake Bay. Loss of Submerged aquatic vegetation results in a loss of food and habitat for many species (US Fish and Wildlife Service, 2011). Of the seven native seagrass species in the Caribbean, two (*Halophila engelmannii* and *H. baillonii*) are considered to be near threatened and vulnerable. Elevated nutrient levels (eutrophication) is the biggest threat in the Americas and is particularly acute in developing nations with rapidly growing economies, where environmental legislation is weak. These local and regional threats exist with a backdrop of environmental change and sea level rise.

Recent trends. There was considerable loss, degradation, and fragmentation of seagrasses, as 2.6 km^2 in Biscayne Bay (Florida, USA) between 1938 and 2009 (Santos et al., 2016). Extensive losses have been reported from Canada (Matheson et al., 2016), and the Caribbean (Van Tussenbroek et al., 2014). The Caribbean Coastal Marine Productivity program found that most study sites showed a decline in seagrass health between 1993 and 2007 (Van Tussenbroek et al., 2014). However, in some areas that have undergone restoration and controls on nutrients, such as Chesapeake Bay in the USA, there has been some recovery (Chesapeake Bay Program, 2017). In cases where nutrient limitations are implemented, recovery is a very slow process, involving the replacement of fast-growing macroalgae with slower-growing plants. Simulation models predict recovery times of several years for fast-growing seagrasses to centuries for slow-growing seagrasses following nutrient reduction (Duarte, 1995).

Coral reefs

Status. Coral reefs are one of the most productive and diverse ecosystems in the world. In addition to the many species of corals, they include populations of sponges, Echinoderms, mollusks such as giant clams, nudibranchs, and octopuses, crustaceans such as crabs, lobsters and shrimp, and a huge diversity of fishes, all of which are either directly or indirectly dependent on the foundation species, the corals. When corals degrade or disappear the rest of the community degrades or disappears. Coral reefs perform vital ecosystem services in tropical countries: they serve as protection against storms, attenuating wave intensity, their fisheries are a source of food for millions of people, and they are a source of considerable revenue from tourism.

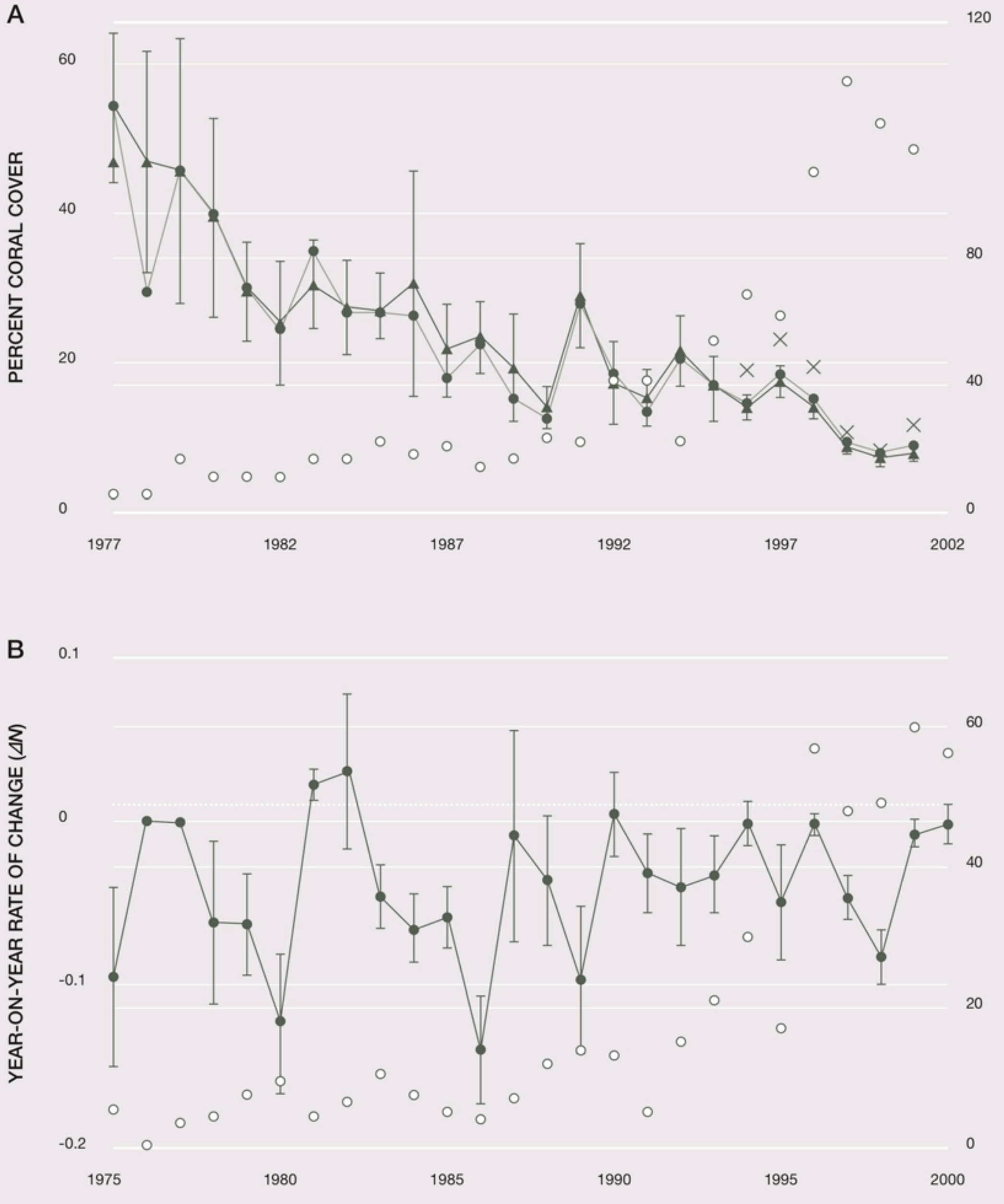
Recent trends. Gardner et al. (2003) found that live coral cover in the Caribbean was reduced from more than 50% in the 1970s to just 10% today (Figure 3.28). This decline was followed by widespread and severe coral bleaching in 2005, which was in turn followed by high coral mortality as a result of disease at many locations. Healthy corals are rare on the intensively studied reefs of the Florida reef tract, USA Virgin Islands and Jamaica (Gardner et al., 2003). Furthermore, two of the formerly most abundant foundation species of Caribbean reefs, the elkhorn coral (*Acropora palmata*) and staghorn coral (*Acropora cervicornis*), have been added to the US Endangered Species List. The decline of herbivorous species (e.g. parrotfish) in coastal marine areas has also been of consequence especially as many are vital to reef resilience (Mumby et al., 2006). Many reef fish continue to be exploited (e.g. endangered *Nassau grouper*, *Epinephelus striatus*)

(Sadovy & Eklund, 1999). Jackson et al. (2014) found that the average coral cover for 88 locations in the Caribbean declined from 34.8% in 1984 to 19.1% in 1998 to 16.3% at the time of the report, but there was great disparity among sites. In contrast, macroalgal cover increased from 7% to 23.6% between 1984 and 1998 and held steady but with even greater disparity among locations since 1998. Differences among locations can be attributed to local factors such as human population density, overfishing of herbivorous fishes, and invasive species. The invasion of the predatory lionfish (see Chapter 1) has been particularly devastating to populations of herbivorous fish. The massive loss of corals in the Caribbean (see Chapter 4 for drivers) has been associated with increases in large seaweeds (macroalgae), outbreaks of coral bleaching and disease, and failure of corals to recover from natural disturbances like hurricanes (Jackson et al., 2014). There are attempts to restore some *Acropora* reefs in the Caribbean with more tolerant strains. Bozec et al. (2016) concluded that reduced fishing for parrotfish and other herbivores would make reefs more resilient to warming.

Global warming is placing Caribbean coastal ecosystems under further stress (see Chapter 4). The predicted increased severity of hurricanes and greater rainfall seasonality here are also likely to increase stress (Fish et al., 2009). In Brazilian reefs of the Southwestern Atlantic Ocean, long-term sea water thermal anomaly events, equal or higher than 1ºC, were responsible for more than 30% of bleached corals in the inshore reefs from 1998 to 2005, (Leão et al., 2010).

Figure 3 28 Total observed change in percent coral cover across the Caribbean basin during the past three decades.

A Percent coral cover from 1977 to 2001. Annual coral cover estimates (▲) are weighted means with 95% bootstrap confidence intervals. Also shown are unweighted mean coral cover estimates for each year (●), the unweighted mean coral cover with the Florida Keys Coral Monitoring Project omitted (×), and the sample size (number of studies) for each year (O). B Year-on-year rate of change [mean ΔN ± SE] in percent coral cover across all sites between 1975 and 2000 (●),which largely fall below the dotted line representing no change, and the number of studies for each period (O). Source: Gardner *et al.* (2003).



3.5 Perils and opportunities for conservation

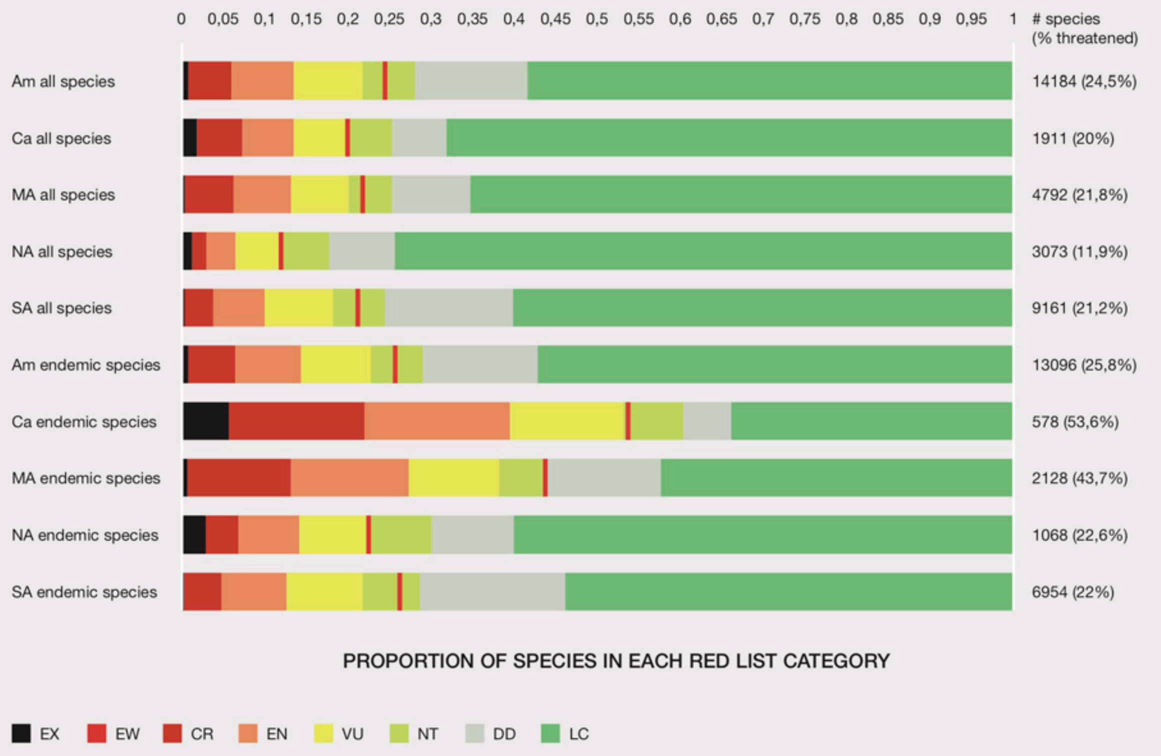
3.5.1 Threat status and temporal trends

Knowledge of threat status, temporal trends, and the main causes underlying extinction probability constitute useful information for policymakers for prioritizing recuperation plans and protection measures and for other stakeholders who wish to reap well-being benefits from particular species or contribute to biodiversity conservation.

Status. Overall, 14,184 species from taxonomic groups within which > 90% of species have been globally assessed by IUCN for extinction risk and synthesized by Brooks et al. (2016) are present in the Americas. Groups assessed cover mammals, birds, chameleons, amphibians, sharks and rays, selected bony fish groups (angelfishes and butterflyfishes, tarpons and ladyfishes, parrotfishes and surgeonfishes, groupers, wrasses, tunas and billfishes, hagfishes, sturgeon, blennies, pufferfishes, seabreams, porgies, picarels), freshwater caridean shrimps, cone snails, freshwater crabs, freshwater crayfish, lobsters, reef-building corals, conifers, cacti, cycads, seagrasses, and plant species occurring in mangrove ecosystems. Conspicuously absent are the majority of flowering plants. Recognizing that available data is strongly skewed towards animals, in total, 24.5% of assessed species are documented as threatened with a high risk of extinction in the wild in the medium term future. The inclusion of data-deficient species for these groups could shift this percentage to as high as 34.7% or as low as 21.2%. The great majority of species assessed for the taxonomic groups mentioned (92.3%, 13,096 species) are endemic to the Americas region.

Figure 3 29 Extinction risk for species in the Americas as a whole (Am) and by subregion (Caribbean: Ca, Mesoamerica: MA, North America: NA, South America: SA).

Red lines show midpoint estimate of proportion of threated species. The top 5 rows are all assessed species in the dataset, and the bottom 5 are the subset of endemic species. Source: Data synthesized by Brooks *et al.* (2016).

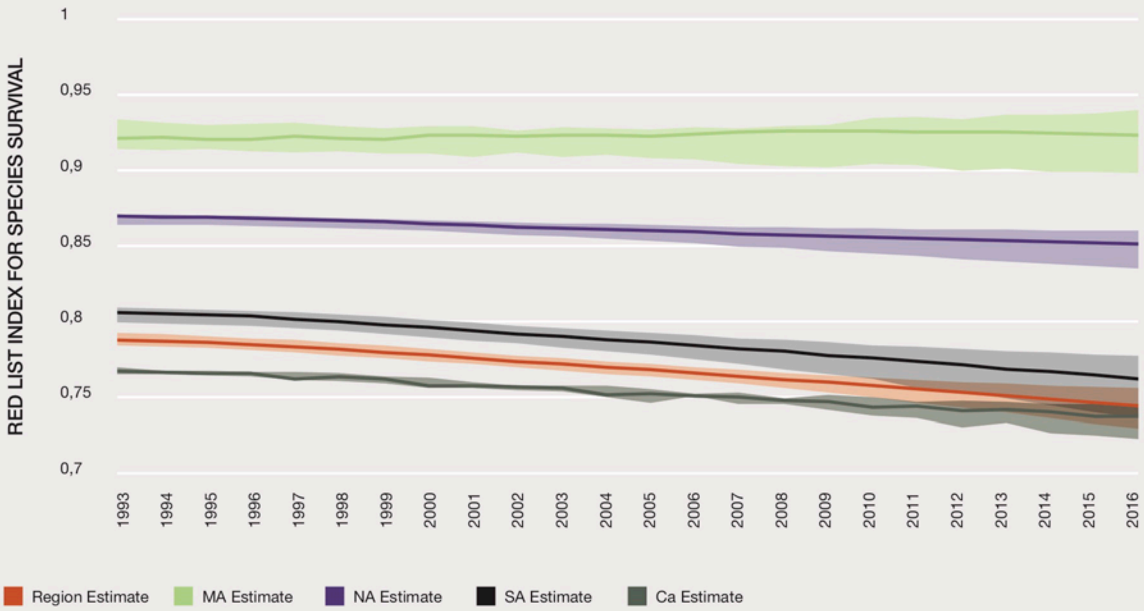


Notable differences in extinction risk characterize the different subregions of the Americas (Figure 3.29). Considering all species, North America shows much lower extinction risk than South America, Mesoamerica, and the Caribbean. With the exception of South America, extinction risk tends to be higher among endemic species. Especially high extinction risks for endemics are found in the Caribbean and Mesoamerica.

Recent trends. For mammals, birds, amphibians, corals, and cycads, global assessments of extinction risk against the Red List categories and criteria have been undertaken multiple times over the last three decades to derive Red List indices as indicators of the rate at which species groups are sliding towards extinction; these can be combined with species distribution data to produce geographically downscaled Red List indices (Rodrigues et al., 2014). According to this criterion, overall the extinction risk has increased over the last 23 years in the Americas, but again there are notable subregional differences (Figure 3.30). Extinction risk in the North America subregion has increased slightly, in Mesoamerica it has remained relatively steady, while in the Caribbean and South American it increased the fastest. Species in the Caribbean region are declining towards extinction the fastest of all but, of course, there are fewer overall species here (Brooks et al., 2016).

Figure 3 30 **Red List indices of species survival for mammals, birds, amphibians, corals, and cycads, weighted by the fraction of each species' distribution occurring within each region/subregion.**

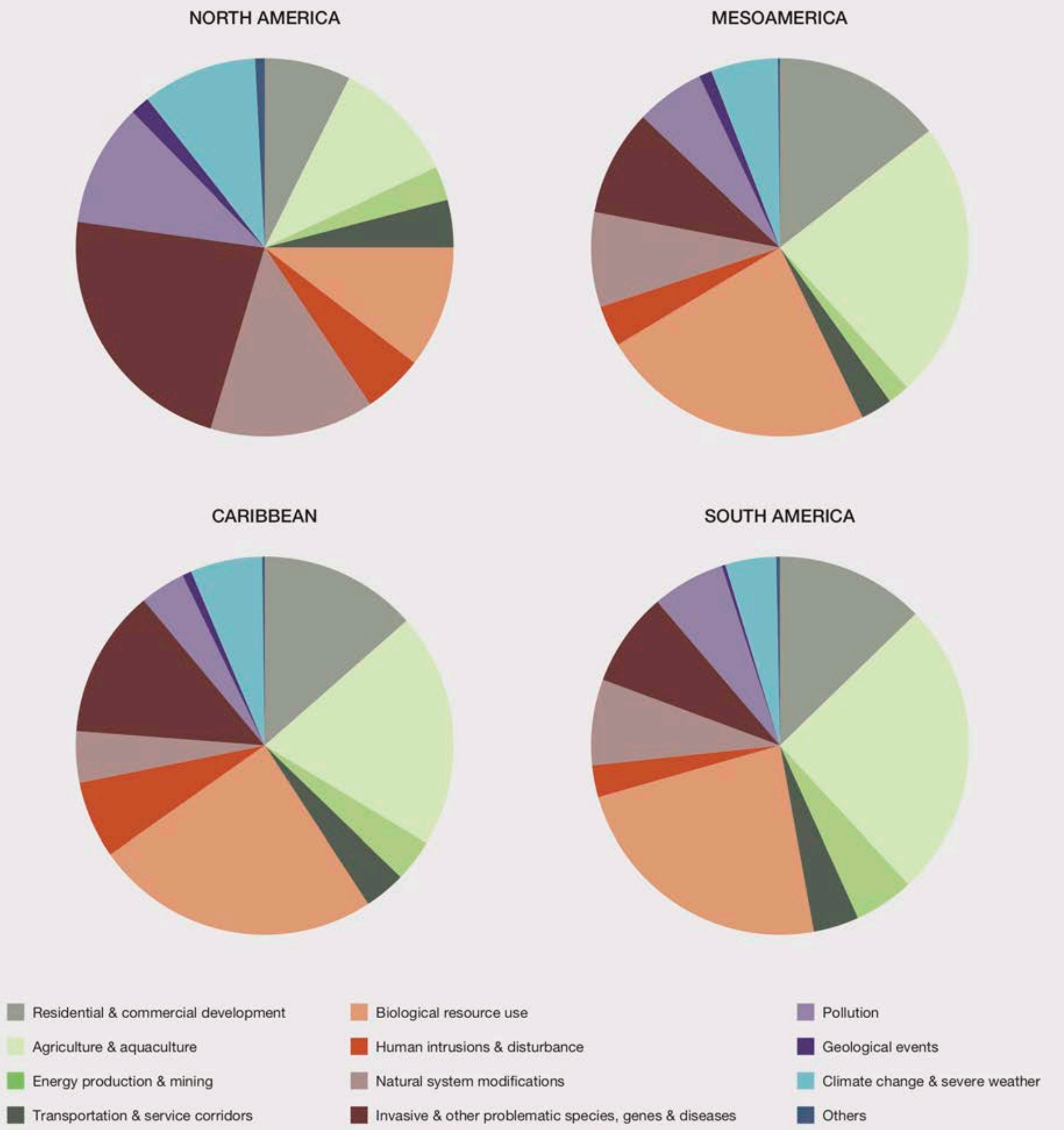
The position on the y-axis indicates the aggregate extinction risk facing species in the region overall. It ranges from 1 (if no species are threatened with extinction) to 0 (if all species are extinct). The horizontal axis shows time, so the slope of the lines for each subregion shows how the extinction risk of the species in that subregion has been changing. A declining slope indicates that extinction risk is growing; an increasing slope indicates that extinction risk is declining. Source: Data synthesized by Brooks *et al.* (2016).



The main threats in the North American subregion come under the IUCN category termed "Invasive & other problematic species (whose origins are uncertain), Genes & diseases" (Figure 3.31). In the other three subregions, the main threats are "Agriculture & aquaculture" and "Biological resource use". While it was seen earlier that there are many alien and invasive species in the Caribbean, the category of "Invasive & other problematic species, Genes & diseases" does not rank high as a threat, at least in the groups assessed to date. The relatively less importance still of the invasive species category in Mesoamerica and South America could relate to the fact that invasive species are less prevalent at tropical latitudes. The overall pattern for these last subregions mirrors the global threat trends (Maxwell et al., 2016). Again, it should be borne in mind that species assessed are strongly skewed towards animal groups. Trends could change measurably with the inclusion of the many threatened plant species in the Americas. Throughout the Americas, biological resource use may be a primary concern in the marine environment (McCauley et al., 2015).

Figure 3 31 **Comparison of the main causes of extinction risk in the Americas.**

When a species is threatened by more than one cause, all causes were included to calculate the proportion. Source: Data from IUCN Red List threat classification, IUCN (2017).

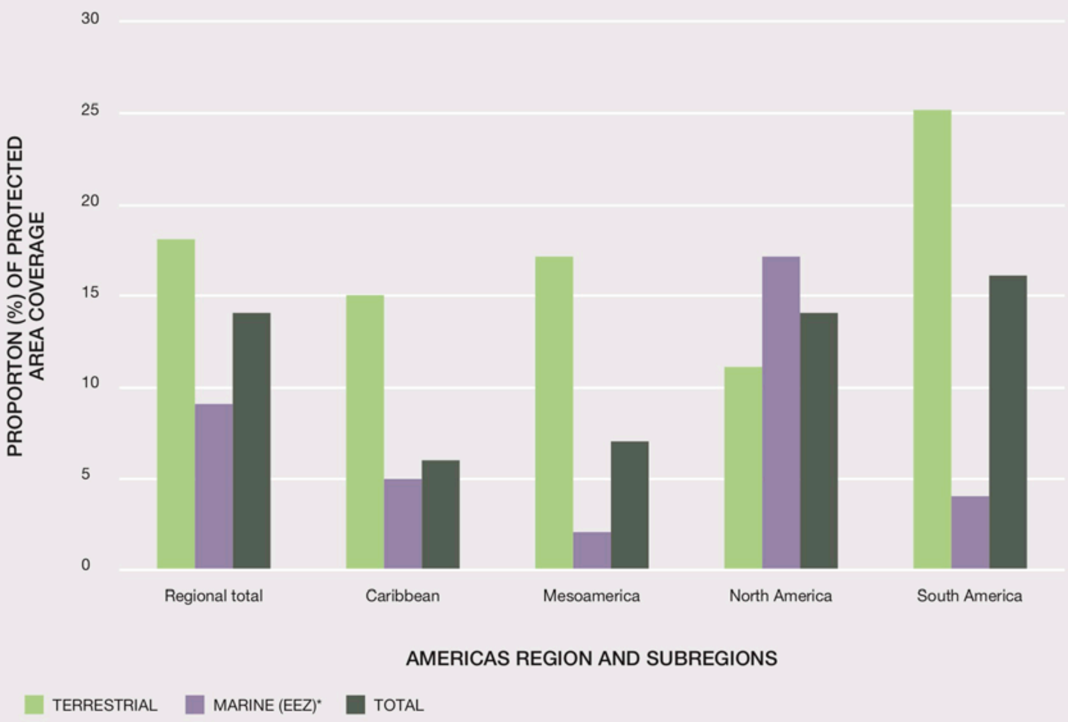


3.5.2 Protected areas

Most early protected areas in the Americas were established with the aim of protecting iconic landscapes. Heightened concern over environmental degradation and the importance of biodiversity led to changes in the motives for establishing protected areas, with an increasingly greater emphasis placed on *in situ* conservation, coverage of KBA (Key Biodiversity Areas), "hotspots", ecosystem services and indigenous rights. Simultaneously, the range of stakeholders involved in establishing protected areas expanded to include private citizens, in addition to governments. Many early protected areas established in mountainous landscapes today perform important roles in protecting key ecosystem services such as water regulation and slope stability.

Status. Total protected area coverage for the Americas is 14%, with 18% of its terrestrial area and 9% of its marine area (within the Exclusive Economic Zone, EEZ) protected (Figure 3.32). Protected area coverage shows variation both among subregions and in relation to the relative amount of land and the marine EEZ protected. For terrestrial habitats, South America has the highest fraction of land in protected areas, whereas for EEZ marine protection, North America has made the most advances (Figure 3.32). Chile recently announced the creation of two new large marine protected areas (around the Juan Fernández Islands and in the Cape Horn-Isla Diego Ramírez area) (Ministerio del Medio Ambiente, 2017) in the South American subregion. Mexico announced the creation of Parque Nacional Revillagigedo (CONANP, 2017) in the Mesoamerican subregion. The Americas, thus are responding rapidly to the challenge of marine protection. These new marine protected areas are not included in Figure 3.32.

Figure 3 32 **Percentage of terrestrial, marine and total protected area coverage in the Americas region and subregions. Source: Based on UNEP-WCMC & IUCN (2015), synthesized by Brooks *et al.* (2016).**



Note: the data for the North American subregion includes protected area in the Hawaiian Islands.

* Economic exclusive zone.

Recent trends. Over the past few decades, there has been an increase in the number of protected areas and the amount of land protected throughout the Americas region (see Chapter 2). In North America, the number of protected areas has almost tripled, and in the Caribbean, it has almost doubled. Protected areas came slowly to Mesoamerica, but have increased in number from 150 to more than 700 since the 1980s, and in South America, they have increased more than four-fold. In South America, over the past 10 years, an additional 683,000 km^2 of new protected areas were added to the Amazon Basin by different countries, increasing the amount of the Amazon protected by 10% (Charity et al., 2016). According to the most recent analysis for terrestrial biomes, a large number of biomes in the Americas are better protected than the global average (Table 3.6); however, despite advances, and of concern given the rapid rate of conversion in many (3.4), some fall well below the global rate. It should be pointed out that the exact level of protection in these biomes is constantly changing because of new initiatives and depends also on how the various biomes are defined, which is far from uniform. In general, it can be seen that closed forests are better protected in relation to the global rate than non-forested areas and wet forests better than dry forests.

Table 3.6. Percentage protection of terrestrial biomes in the Americas according to biogeographic realm. The North American realm (= Nearctic realm in Jenkins & Joppa, 2009) extends into Mexico and thus is larger than the corresponding IPBES subregion. The Neotropical realm includes South American and Caribbean subregions and part of the Mesoamerican subregion as defined by the IPBES. Biomes shown in bold enjoy a high level of protection relative to the global rate in at least one of the biogeographical realms. Based on data in Jenkins & Joppa (2009).

Biome	Global	North American	Neotropical
Tropical and subtropical moist broadleaf forests	21		**32**
Tropical and subtropical dry broadleaf forests	8	0	9
Tropical and subtropical coniferous forests	7	7	8
Temperate broadleaf and mixed forests	11	12	**29**
Temperate coniferous forest	25	**33**	
Boreal forests/taiga	9	10	
Tropical and subtropical grassland, savannas and shrubland	13	8	11
Temperate grasslands and savannas	4	3	2
Flooded grasslands and savannas	20		15
Montane grasslands and shrublands	25		14
Tundra	17	**22**	
Mediterranean forests, woodland and scrub	7	**21**	1
Deserts and xeric vegetation	9	**14**	9
Mangroves	21		**37**

With regard to priority areas for conservation, the Americas region hosts 20% of globally identified KBA (Table 3.7). KBA include the 12,000 Important Bird and Biodiversity Areas (IBAs), identified by BirdLife International (2015), plus Alliance for Zero Extinction (AZE) sites (Ricketts et al., 2005) and other KBA identified through hotspot profiles supported by Critical Ecosystem Partnership Fund (World Database of Key Biodiversity Areas, n.d.).

Table 3.7. Number and percentage of KBA by subregion in the Americas relative to the global total.

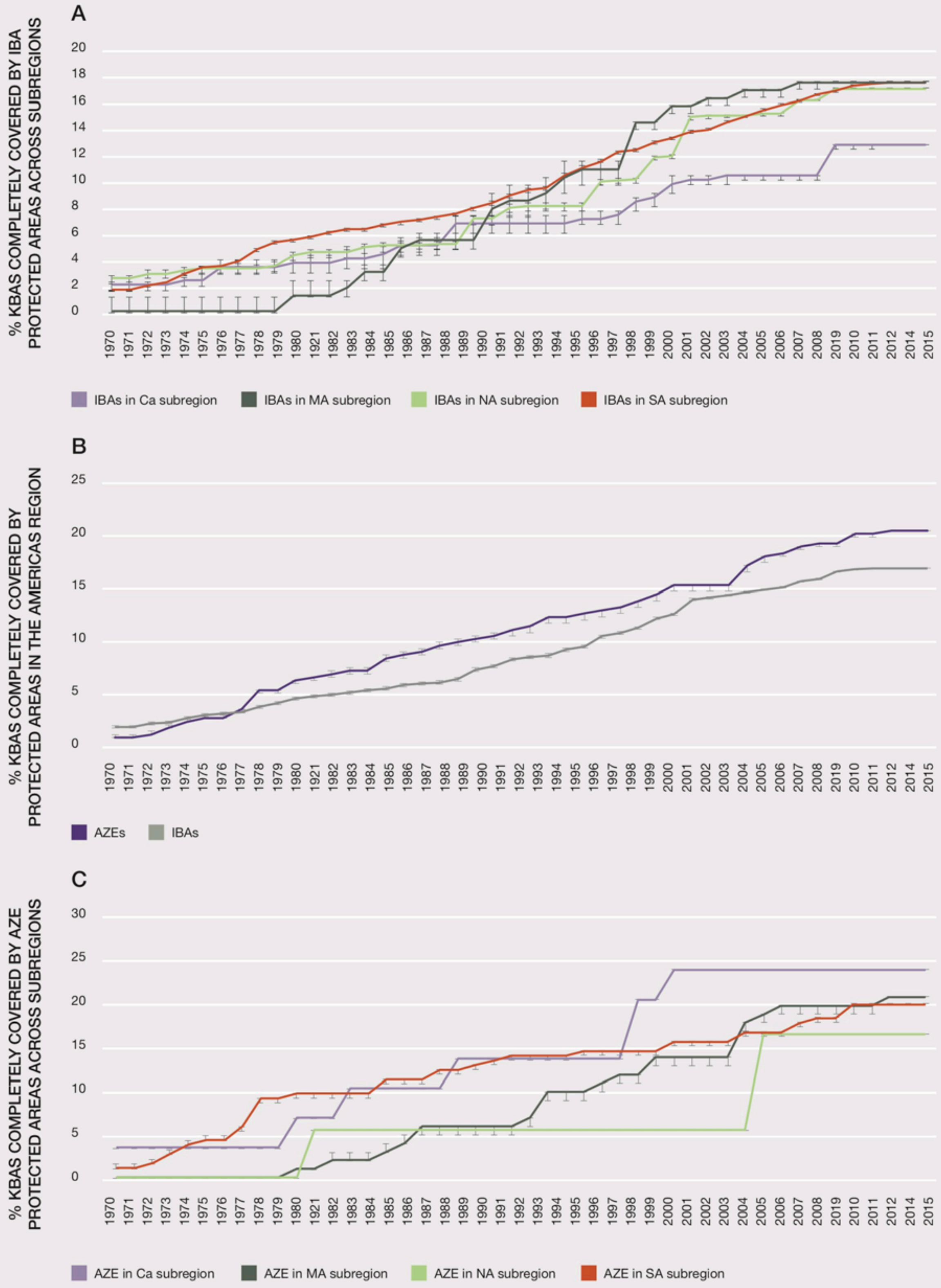
Region	# KBA	%
North America	985	6.35
Caribbean	419	2.70
Mesoamerica	305	1.96
South America	1,371	8.83
Americas	3,080	19.84
Global	15,524	100

Source: Data are from the World Database of Key Biodiversity Areas™, searched October 22, 2017. http://www.keybiodiversityareas.org/ site/search.

The total protected area coverage of KBA has increased significantly over the past 50 years (Figure 3.33). Brooks et al. (2016) synthesize all three datasets for the Americas region. Currently (as of 2015) 17.0% of IBAs and 20.6% of AZE sites are fully covered in the Americas as a whole. At the subregional level, for IBAs South America lags strongly behind; for AZE sites the Caribbean takes the lead, while North America lags behind the most (Figure 3.33).

Figure 3 33 **Growth in the proportion of KBAs (Key Biodiversity Areas) completely covered by protected areas in the Americas between 1970 and 2015.**

A Trends in the four American subregions for IBAs (Important Bird and Biodiversity Areas). B Trends in the four American subregions for AZEs (Zero Extinction sites). C Trends in the Americas as whole for both IBAs and AZEs. Source: IUCN & Birdlife International (2016) as synthesized by Brooks *et al.* (2016).



With the increasing recognition of indigenous rights and public recognition of NCP, the establishment of indigenous and private reserves has increased notably. Indigenous reserves in South America tend to be concentrated in tropical forests where they contribute greatly to the integrity of ecosystem services, and the sustainable use of many plant and animal species used for human-well-being. Currently, indigenous reserves in Latin America and the Caribbean account for around 12% of all protected land (Nelson & Chomitz, 2011) (for more details on the contribution of indigenous reserves to human well-being see Chapter 2). In the Amazon, around 3000 indigenous lands (not all recognized) now cover over 2 million km^2 (Charity et al., 2016; Figure 3.34). Both uninhabited protected areas (parks) and indigenous lands have proven to reduce deforestation and fire in South American wet tropical forest (Armenteras et al., 2009; Nepstad et al., 2006; Nelson & Chomitz, 2011), and contain viable populations of most threatened tree species (ter Steege et al., 2015).

Figure 3 34 **Location of protected indigenous lands, other indigenous lands and protected areas in the Amazon basin as of March 2016. Source: Charity *et al.* (2016).**

Private conservation efforts are now important in the temperate forests of southern South America (Pliscoff & Fuentes-Castillo, 2011), the Mediterranean forests, woodland and scrub biome in California (Paulich, 2010), and in Brazil in general (de Vasconcellos Pegas & Castley, 2016). Brazil's private reserves are distributed across seven biomes (six terrestrial and the marine); they are recognized under federal law and created to protect nature in perpetuity. Private conservation efforts in the USA have been stimulated by the fact that around two-thirds of the land in the continental USA is privately owned and three-quarters of all threatened or endangered species depend on private land for habitat, food or breeding (Paulich, 2010). A similar situation could occur in the South American Mediterranean biome. While private initiatives are noteworthy, they sometimes risk outcomes of the establishment of protected areas in places that are large and cheap but of less importance for biodiversity conservation (Barnes, 2015), or choices being made on purely aesthetic grounds increasing protection where it sometimes is perhaps less required. It is therefore essential to complement these measures with measures of safeguard of important sites (Butchart et al., 2016) and encourage protection where it is most needed, regardless of aesthetic value.

Despite the overall increase in protection and notable conservation success stories (e.g. Carabias et al., 2010), major conservation incongruencies within many biomes still remain. Incongruencies address both what and how much is conserved. With respect to what is conserved, as an example, although California has pioneered multiple species habitat conservation plans and other regional and multi-benefit approaches to enhance integrated planning of protected areas (Pincetl et al., 2016), unprotected areas tend to harbor the highest numbers of rare plant taxa (Pavlik & Skinner, 1994), while important areas with high levels of plant neoendemism fall outside of protected lands (Kraft et al., 2010). How common this trend is in other biomes remains to be seen and should be a priority question.

With respect to how much is conserved, as examples, the Central American system of protected areas currently includes 669 protected areas summing 129,640 km^2, the majority of which correspond to moist tropical and subtropical forest (Programa Estado de la Nacion, 2008; The Nature Conservancy, 2005). For Mesoamerica defined as the five southernmost states of Mexico to the Darien in eastern Panama, while 29% of tropical broad-leaved forest is protected, only 10% of coniferous forest comes under protection (DeClerck et al., 2010). For South American moist tropical and subtropical forests, less than 2% of Atlantic rainforest is protected.

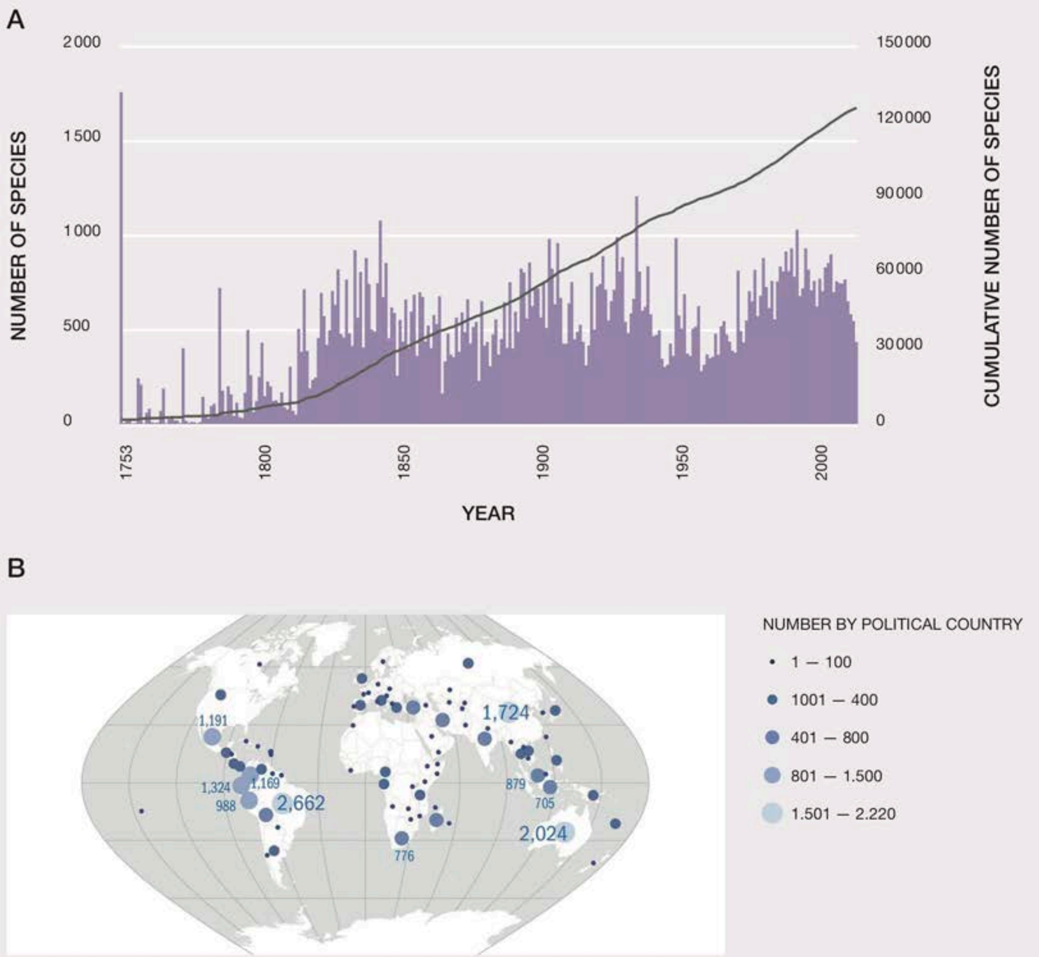
Incongruencies are even more extreme in other biomes. Overall, only 0.3% of Tropical dry forest in Mesoamerica, 7% in South America and 10% in the Caribbean is protected (Portillo-Quintero & Sánchez-Azofeifa, 2010); this percentage descends to 0.2% in Mexico and 1.0% in Venezuela, but is a much higher 15% in Costa Rica (Portillo-Quintero & Sánchez-Azofeifa, 2010), indicating notable differences in individual country efforts. Protection of Chaco is about 10% (Fehlenberg et al., 2017), ranging from 36% in Bolivia to 6.5% in Paraguay. Although the amount of protected land tripled in the wider Mediterranean biome in South America between 1975 and 2017, less than 3% is currently protected (based on data in "http://www.mma.gob.cl") with some particular ecosystems of the biome totally lacking protection (Pliscoff & Fuentes Castillo, 2011). Currently, 8.3% of the Brazilian Cerrado is considered to be under some kind of protection, with only 3.1% in strictly protected areas (National Database for Protected Areas/Brazilian Ministry of the Environment - Cadastro Nacional de Unidades de Conservação - CNUC, updated February 7, 2017). South American drylands are very poorly protected – 1% of land area of the Caatinga (Banda-R et al., 2016; de Oliveira et al., 2012), and 1–2% of Chilean western desert (Arroyo & Cavieres, 1997; Luebert & Pliscoff, 2006). Likewise, in the EEZ much variation is found for marine conservation (Watson et al., 2014). All these incongruencies have many sources, but one obvious one is a lack of systematic planning among countries where a given biome is found.

3.6 Knowledge and data gaps

Biodiversity inventories. Basic inventorying of biodiversity is far from complete in the Americas. Accumulated species descriptions for vascular plants have not yet reached an asymptote (Figure 3.35a). Over the period 2004-2016, Brazil registered the largest number of new plant species names in the International Plant Names Index worldwide (Figure 3.35b). Over 2,000 new species of plants and vertebrates have been described from the Amazon alone since 1999 (Charity et al., 2016). Even in well-known groups such as mammals, 42% of the new species described worldwide between 1993 to 2008 came from the Americas (Ceballos & Ehrlich, 2009), mostly from Mesoamerica and South America. These trends are likely to be repeated for other taxonomic groups. Knowledge of invertebrates is particularly deficient including for taxonomic groups of particular importance for human well-being, such as bees. This assessment has shown that high-quality information on species richness across the entire Americas is available for a very limited number of taxonomic groups. Some estimates of biodiversity, of course, might be exaggerated if care was not taken to remove synonyms. Overall, an accurate estimate of the total biodiversity in the Americas is currently not possible, and is unlikely to become available for a long time at the current rate of progress. Also, systematized knowledge on the use of biodiversity is still scarce, despite major efforts made in Mexico, Costa Rica, Brazil, and Colombia.

Figure 3 35 **Sources of new vascular plant species names entered into the International Plant Names Index.**

A The number of plant species (basionyms) described per year from 1753 to 2015 for the Americas, and the cumulative number of accepted species.
B Sources of new vascular plant species names entered into the International Plant Names Index between 2004 and 2016 for different countries. Source: Willis (2016) Original data as in updated for the years 2004 to 2016, Ulloa Ulloa *et al.* (2017).

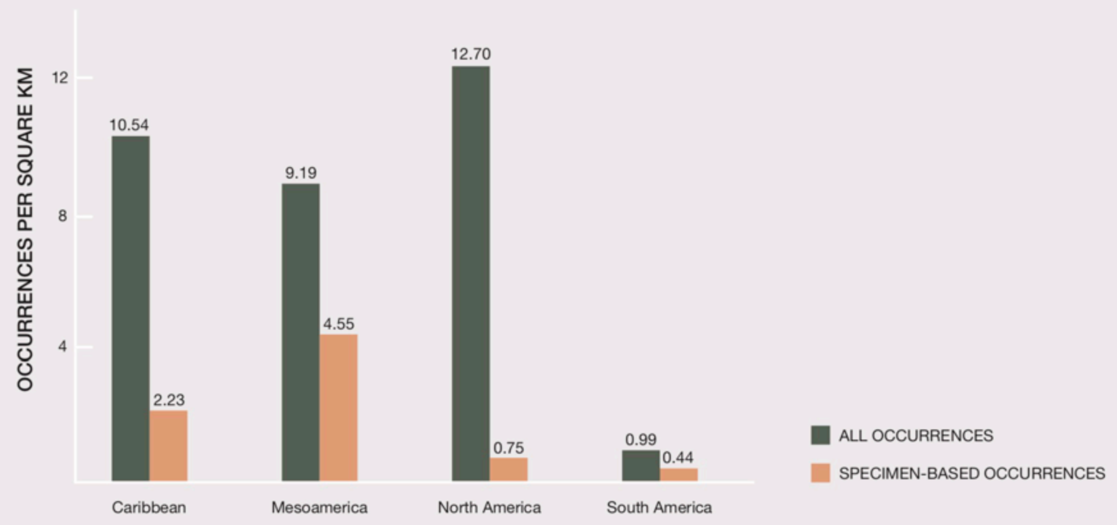


Similar and probably even much larger knowledge gaps occur in the marine (and probably freshwater) realms. Based on their studies, it is predicted that only about half of marine organisms have been described for the Atlantic and Pacific coasts of South America (Miloslavich et al., 2011); as on the land, a severe lack of taxonomic expertise in the subregion is a major handicap.

Mobility of biodiversity data. Progress in the detection of the impacts of climate change on biodiversity, conservation gaps, and areas with high concentrations of invasive species today depends heavily on georeferenced biodiversity occurrence data. Overall, 50% of georeferenced online occurrence data in the Global Biodiversity Information Facility pertains to the Americas. However, the density of georeferenced data varies widely among subregions (Figure 3.36) (and between countries within each subregion – not shown). Causes include differences in intrinsic richness among countries, a greater level of collaboration between foreign institutions and the tropical countries, differences in exploration intensity, lack of manpower to digitalize biodiversity data and some reticence still on the part of some institutions to incorporate their biodiversity data into the Global Biodiversity Information Facility. The South American subregion lags behind, but important efforts are getting underway. For example, specimens from several institutions in Argentina, thanks to support by the Argentinian National Science Council, can now be found in the Global Biodiversity Information Facility. Brazil is creating the Brazilian Information System on Biodiversity and the "*Portal da Biodiversidade*" which are first steps to consolidate biodiversity data and make it available online. The Chilean national science council is contemplating making it compulsory for grant-holders to place biodiversity data collected with national research funds in the Global Biodiversity Information Facility.

Figure 3 36 **Quantity of georeferenced biodiversity occurrences in the Global Biodiversity Information Facility for subregions of the Americas divided by land area.**

Note: Occurrence data for North America includes Hawaii. Source: GBIF (http://www.gbif.org/occurrence). Data accessed: March 26, 2017.



Importantly, efforts are being made to build comprehensive alien species databases at the country (e.g. USA, Brazil, Mexico, Chile) and regional (e.g. Invasives Information Network) levels. Not having access to all biodiversity data, in addition to hindering research progress, introduces uncertainty in the results of regional and global-scale studies that rely heavily on occurrence data and lowers the quality of environmental impact studies within countries.

Biome and ecosystem-level data. With very few exceptions, we currently lack accurate knowledge of biodiversity at the biome level. Where available, the information is limited to a few groups of better-known organisms and does not necessarily coincide with the spatial delineation of the World Wildlife Fund terrestrial biomes adopted by the assessment (see Chapter 1). These have been major obstacles in this assessment. Overall, studies, when present, are insufficient in number for performing biome-level meta-analyses. Thus the assessments of the units analysis in Chapter 3 are necessarily descriptive and piecemeal. Revision of the World Wildlife Fund biomes based on a consensus is highly desirable now that more accurate vegetation mapping is possible and can be combined with verified species distribution data. If all countries were to adopt such a system, this would be an enormous step forward. One reason for a lack of biome-level data is that many biomes in the Americas cross country boundaries. For example, high elevation systems in South American are found in seven countries and span about 44 degrees of latitude, Mesoamerican dry tropical forest stretches over seven countries and the Amazonian basin over eight. This transnational problem is far less acute in the North American subregion composed of only three countries. Because governments are usually first concerned with the biodiversity of their respective countries, resources for undertaking cross-country, biome-level surveys are generally lacking, but of course, this is not the only reason. This represents a serious challenge for future regional and global IPBES assessments and undermines the efficiency of conservation measures in biomes.

Data on population sizes and genetic diversity is scarce outside the North American subregion. Likewise, long-term series data are few and far between making it difficult to detect temporal trends. Throughout the Americas, fishes and invertebrates differ in their population status, yet the exploitation status of many species is unknown across several taxonomic groups, in particular, elasmobranchs (sharks, skates and rays) and coastal fishes because of a lack of long-term series data. For terrestrial habitats, in the early 1990s, pioneering efforts in the US Long-Term Ecological Research Network led to the International Long Term Ecological Research Network (Vanderbilt & Gaiser, 2017). Although many formally accredited sites are found in the Americas, these are strongly concentrated in the USA, Mexico, and Brazil. There are no high altitude International Long Term Ecological Research sites along the entire length of the high Andes where global warming is occurring faster than in adjacent lowlands. Nevertheless, the GLORIA program (www.mountainstudies.org/climate-change) has been active in setting up monitoring sites in the northern and central Andes, to be extended now to the southern Andes. For the marine domain, two North American marine sites were recently accredited by International Long Term Ecological Research Network.

Biodiversity-ecosystem functions-NCP linkages. Most work in this area in the Americas comes from the North American subregion and has involved plot-based studies with a strong focus on productivity. Some information exists in the agricultural, fisheries, pollination, and hydrological domains in the other subregions. Across the Americas, vascular plants comprise the only taxonomic group for which the coverage of functional trait data is abundant (Kattge et al., 2011), yet gaps in functional trait data are highest precisely where diversity tends to be highest: i.e., tropical latitudes (Jetz et al., 2016). Studies linking biodiversity and other less tangible kinds of NCP are incipient throughout. The health benefits of biodiversity and level of equity in terms of access to green areas in urban areas, for example, are fairly open fields. A major gap in our understanding, perhaps with the exception of carbon storage, are links between biodiversity and ecosystem services or NCP at large spatial scales. This requires replicated information across individual biomes/units of analysis and hence coordinated research, often in several countries. To advance in our knowledge here, also, greater collaboration between the traditional biodiversity research community and other disciplines is desirable. Two major challenges for the future in the Americas are to standardize information and to make it available in a template that is usable by decision makers. In this sense, initiatives such as the Biodiversity Indicators Partnership (https://www.bipindicators.net/), which make suites of global indicators available to support national-level reporting and/or National Biodiversity Strategies and Action Plans updating and implementation, are promising.

3.7 Concluding remarks

Biodiversity is linked to ecosystem functions and is highly relevant to NCP across the ecologically diverse and species-rich Americas. All units of analysis of the Americas considered contribute to human well-being. However, Tropical and subtropical moist forests, Temperate and boreal forests and woodlands, Tropical and subtropical dry forests, Mediterranean forests, woodlands and scrub, and Tundra and high elevation habitats stand out as particularly critical for NCP delivery. For aquatic systems, freshwater is considered somewhat more important for NCP than marine. Except in a limited number of cases, this chapter shows that the biodiversity in the Americas´ terrestrial biomes and freshwater and marine habitats continues to undergo serious erosion. The introduction and spread of alien species can be expected to continue causing direct and indirect impacts on human well-being and biodiversity. The subregions currently undergoing most dramatic land use change, considering their spatial extent, are South America and Mesoamerica, where conversion of vegetation to support pastures, agriculture and exotic plantation forestry is widespread. These changes are leading to major losses of habitat with concomitant population and species declines. In the marine and freshwater realms, the number of threatened species is high, and many fish species are over-exploited.

Climate change has begun to affect the distribution of biodiversity, but to a greater degree in North America than South America for the moment. Increased fire frequency in several biomes constitutes a growing threat. Despite significant progress in developing protective measures for the land and in the sea, they are often insufficient. The greatest challenges to policymakers and decision makers will be to: arrest or slow habitat loss; encourage more ecologically-friendly management practices to ensure long-term food- and water-security; and promote alternative biodiversity-based economic activities that are less destructive than current activities. These are not new challenges. Progress necessarily implies a conscious, collective societal effort. Many lessons can be learned from indigenous peoples who have succeeded in living in harmony on the land.

3.8 References

Aber, A., & Ferrari, G. (2010). *Lineamientos para la gestión nacional de especies exóticas invasoras*. Montevideo, Uruguay: Comité Nacional de Especies Exóticas Invasoras, DINAMA (Dirección Nacional de Medio Ambiente), and UNESCO. Retrieved from http://unesdoc.unesco.org/images/0019/001906/190691s.pdf

Ackerly, D., Stock, W. D., & Slingsby, J. (2014). Geography, climate, and biodiversity: the history and future of Mediterranean-type ecosystems. In N. Allsopp, J. F. Colville, & G. A. Verboom (Eds.), *Fynbos: Ecology, Evolution, and Conservation of a Megadiverse Region* (pp. 361–375). Oxford, UK: Oxford University Press.

Ackerman, D., & Breen, A. (2016). Infrastructure development accelerates range expansion of trembling aspen (*Populus tremuloides*, Salicaceae) into the Arctic. *Arctic*, *69*(2), 130–136. http://doi.org/10.14430/arctic4560

Adams, C. D. (1997). Regional overview: Caribbean islands. In S. D. Davis, V. H. Heywood, O. Herrera-Macbryde, J. Villa-Lobos, & A. C. Hamilton (Eds.), *Centres of Plant Diversity. A Guide and Strategy for their Conservation. Vol. 3. The Americas.* (pp. 233–258). Cambridge, UK: IUCN and WWF.

Agard, J., Kishore, R., & Bayne, B. (1992). *Perna viridis* (Linnaeus, 1758): first record of the Indo-Pacific green mussel (Mollusca: Bivalvia) in the Caribbean. *Caribbean Marine Studies*, *3*, 59–60.

Aguirre-Muñoz, A., & Mendoza, R. (2009). Especies exóticas invasoras: impactos sobre las poblaciones de flora y fauna, los procesos ecológicos y la economía. In R. Dirzo, R. González, & I. J. March (Eds.), *Capital Natural de México, Vol 2: Estado de Conservación y Tendencias de Cambio.* (pp. 277–318). Mexico City, Mexico: CONABIO.

Ahlström, A., Raupach, M. R., Schurgers, G., Smith, B., Arneth, A., Jung, M., Reichstein, M., Canadell, J. G., Friedlingstein, P., Jain, A. K., Kato, E., Poulter, B., Sitch, S., Stocker, B. D., Viovy, N., Wang, Y. P., Wiltshire, A., Zaehle, S., & Zeng, N. (2015). The dominant role of semi-arid ecosystems in the trend and variability of the land CO2 sink. *Science*, *348*(6237), 895–899. https://doi.org/10.1126/science.aaa1668

Aide, T. M., Zimmerman, J. K., Pascarella, J. B., Rivera, L., & Marcano-Vega, H. (2000). Forest regeneration in a chronosequence of tropical abandoned pastures: Implications for restoration ecology. *Restoration Ecology*, *8*(4), 328–338. http://doi.org/10.1046/j.1526-100x.2000.80048.x

Aitken, D., Rivera, D., Godoy-Faúndez, A., & Holzapfel, E. (2016). Water scarcity and the impact of the mining and agricultural sectors in Chile. *Sustainability*, *8*(2), 128. http://doi.org/10.3390/su8020128

Aizen, M. A., Morales, C. L., Vázquez, D. P., Garibaldi, L. A., Sáez, A., & Harder, L. D. (2014). When mutualism goes bad: density-dependent impacts of introduced bees on plant reproduction. *New Phytologist*, *204*(2), 322–328. http://doi.org/10.1111/nph.12924

Alaska Department of Fish and Game (ADF&G). (2016). Hunting, Trapping, and Shooting. Retrieved January 1, 2016, from http://www.adfg.alaska.gov/index.cfm?adfg=hunting.main

Alayo, P. (1974). Los hemípteros acuáticos de Cuba. *Nueva Serie Torreia*, *36*, 9–64.

Alayón García, G. (1999). Biodiversidad de las arañas (Arachnida: Araneae): estado del conocimiento en Cuba. *Cocuyo*, *8*, 3–8.

Aldunate, C., Villagrán, C., Armesto, J. J., & Castro, V. (1983). Ethnobotany of pre-altiplanic community in the Andes of northern Chile. *Economic Botany*, *37*(1), 120–135. http://doi.org/10.1007/BF02859312

Alexander, J. M., Lembrechts, J. J., Cavieres, L. A., Daehler, C., Haider, S., Kueffer, C., Liu, G., McDougall, K., Milbau, A., Pauchard, A., Rew, L. J., & Seipel, T. (2016). Plant invasions into mountains and alpine ecosystems: current status and future challenges. *Alpine Botany*, *126*(2), 89–103. https://doi.org/10.1007/s00035-016-0172-8

Alho, C. J. R. (2011b). Biodiversity of the Pantanal: its magnitude, human occupation, environmental threats and challenges for conservation. *Brazilian Journal of Biology*, *71*(1 Suppl 1), 229–232. http://doi.org/10.1590/S1519-69842011000200001

Alho, C. J. R., Camargo, G., & Fischer, E. (2011a). Terrestrial and aquatic mammals of the Pantanal. *Brazilian Journal of Biology*, *71*(1 Suppl 1), 297–310. http://doi.org/10.1590/S1519-69842011000200009

Alho, C. J. R., Fischer, E., Oliveira-Pissini, L. F., & Santos, C. F. (2011). Bat-species richness in the Pantanal floodplain and its surrounding uplands. *Brazilian Journal of Biology*, *71*(1 suppl 1), 311–320. http://doi.org/10.1590/S1519-69842011000200010

Allen, C. D., Macalady, A. K., Chenchouni, H., Bachelet, D., McDowell, N., Vennetier, M., Kitzberger, T., Rigling, A., Breshears, D. D., Hogg, E. H., Gonzalez, P., Fensham, R., Zhang, Z., Castro, J., Demidova, N., Lim, J.-H., Allard, G., Running, S. W., Semerci, A., & Cobb, N. (2010). A global overview of drought and heat-induced tree mortality reveals emerging climate change risks for forests. *Forest Ecology and Management*, *259*(4), 660–684. https://doi.org/10.1016/j.foreco.2009.09.001

Almeida-Neto, M., Prado, P. I., Kubota, U., Bariani, J. M., Aguirre, G. H., & Lewinsohn, T. M. (2010). Invasive grasses and native Asteraceae in the Brazilian Cerrado. *Plant Ecology*, *209*(1), 109–122. http://doi.org/10.1007/s11258-010-9727-8

Almeida, W. R., Lopes, A. V., Tabarelli, M., & Leal, I. R. (2015). The alien flora of Brazilian Caatinga: deliberate introductions expand the contingent of potential invaders. *Biological Invasions*, *17*(1), 51–56. http://doi.org/10.1007/s10530-014-0738-6

Alonso, A., Muñoz-Carpena, R., Kennedy, R. E., & Murcia., C. (2016). Wetland landscape spatio-temporal degradation dynamics using the new Google Earth engine cloud-based platform: Opportunities for non-specialists in remote sensing. *Transactions of the ASABE*, *59*(5), 1331–1342. http://doi.org/10.13031/trans.59.11608

Altieri, M. A. (2000). Developing sustainable agricultural systems for small farmers in Latin America. *Natural Resource Forum*, *24*(2), 97–105. http://doi.org/10.1111/j.1477-8947.2000.tb00935.x

Altieri, M. A. (2009). The ecological impacts of large-scale agrofuel monoculture production systems in the Americas. *Bulletin of Science, Technology & Society*, *29*(3), 236–244. http://doi.org/10.1177/0270467609333728

Alvarez, R. (2001). Estimation of carbon losses by cultivation from soils of the Argentine Pampa using the Century Model. *Soil Use and Management*, *17*(2), 62–66. http://doi.org/10.1079/SUM200165

Alvarez, R. (2005). Carbon stocks in pampean soils: a simple regression model for estimation of carbon storage under nondegraded scenarios. *Communications in Soil Science and Plant Analysis*, *36*(11–12), 1583–1589. http://doi.org/10.1081/CSS-200059082

Anacker, B. L. (2014). The nature of serpentine endemism. *American Journal of Botany*, *101*(2), 219–224. http://doi.org/10.3732/ajb.1300349

Anadón-Irizarry, V., Wege, D. C., Upgren, A., Young, R., Boom, B., León, Y. M., Arias, Y., Koenig, K., Morales, A. L., Burke, W., Pérez-Leroux, A., Levy, C., Koenig, S., Gape, L., & Moore, P. (2012). Sites for priority biodiversity conservation in the Caribbean Islands biodiversity hotspot. *Journal of Threatened Taxa*, *4*(8), 2806–2844. https://doi.org/10.11609/JoTT.o2996.2806-44

Anderson-Teixeira, K. J., Snyder, P. K., Twine, T. E., Cuadra, S. V., Costa, M. H., & DeLucia, E. H. (2012). Climate-regulation services of natural and agricultural ecoregions of the Americas. *Nature Climate Change*, *2*(3), 177–181. http://doi.org/10.1038/nclimate1346

Anderson, P. K., Cunningham, A. A., Patel, N. G., Morales, F. J., Epstein, P. R., & Daszak, P. (2004). Emerging infectious diseases of plants: Pathogen pollution, climate change and agrotechnology drivers. *Trends in Ecology and Evolution*, *19*(10), 535–544. http://doi.org/10.1016/j.tree.2004.07.021

Andréassian, V. (2004). Waters and forests: from historical controversy to scientific debate. *Journal of Hydrology*, *291*(1), 1–27. http://doi.org/10.1016/j.jhydrol.2003.12.015

Andresen, C. G., & Lougheed, V. L. (2015). Disappearing Arctic tundra ponds: fine-scale analysis of surface hydrology in drained thaw lake basins over a 65 year period (1948–2013). *Journal of Geophysical Research: Biogeosciences*, *120*(3), 466–479. http://doi.org/10.1002/2014JG002778

Antunes, A. P., Fewster, R. M., Venticinque, E. M., Peres, C. A., Levi, T., Rohe, F., & Shepard, G. H. (2016). Empty forest or empty rivers? A century of commercial hunting in Amazonia. *Science Advances*, *2*(10), e1600936. http://doi.org/10.1126/sciadv.1600936

Aponte, H., & Cano, A. (2013). Estudio florístico comparativo de seis humedales de la costa de Lima (Perú): actualización y nuevos retos para su conservación. *Revista Latinoamericana de Conservación*, *3*(2), 15–27.

Aravena, J. C., Lara, A., Wolodarsky-Franke, A., Villalba, R., & Cuq, E. (2002). Tree-ring growth patterns and temperature reconstruction from *Nothofagus pumilio* (Fagaceae) forests at the upper tree line of southern Chilean Patagonia. *Revista Chilena de Historia Natural*, *75*(2), 361–376. http://doi.org/10.4067/S0716-078X2002000200008

Arbeláez-Cortés, E., Marín-Gómez, O. H., Baena-Tovar, O., & Ospina-González, J. C. (2011). Aves, Finca Estrella de Agua - Páramo de Frontino, municipality of Salento, Quindío, Colombia. *Check List*, *7*(1), 064–070. http://doi.org/10.15560/7.1.64

Arias-González, J. E., Rodríguez-Peña, O. N., Almeida-Leñero, L., Hernández Almeida, O. U., & Schmitter-Soto, J. J. (2016). Cambios en la biodiversidad y sus consecuencias en el funcionamiento de los ecosistemas y sus servicios. In P. Balvanera, E. Arias-González, R. Rodríguez-Estrella, L. Almeida-Leñero, & J. J. Schmitter-Soto (Eds.), *Una Mirada al Conocimiento de los Ecosistemas de México* (pp. 191–228). Mexico City, Mexico: Universidad Nacional Autónoma de México.

Arino, O., Ramos Perez, J. J., Kalogirou, V., Bontemps, S., Defourny, P., & Van Bogaert, E. (2012). Global Land Cover Map for 2009 (GlobCover 2009). European Space Agency (ESA), Université catholique de Louvain (UCL), PANGAEA. http://doi.org/10.1594/PANGAEA.787668

Armas, C., Gutiérrez, J. R., Kelt, D. A., & Meserve, P. L. (2016). Twenty-five years of research in the north-central Chilean semiarid zone: The Fray Jorge Long-Term Socio-Ecological Research (LTSER) site and Norte Chico. *Journal of Arid Environments*, *126*, 1–6. http://doi.org/10.1016/j.jaridenv.2015.12.008

Armenteras, D., Gast, F., & Villareal, H. (2003). Andean forest fragmentation and the representativeness of protected natural areas in the eastern Andes, Colombia. *Biological Conservation*, *113*(2), 245–256. http://doi.org/10.1016/S0006-3207(02)00359-2

Armenteras, D., Rodríguez, N., & Retana, J. (2009). Are conservation strategies effective in avoiding the deforestation of the Colombian Guyana Shield? *Biological Conservation*, *142*(7), 1411–1419. http://doi.org/10.1016/j.biocon.2009.02.002

Armenteras, D., & Rodríguez Eraso, N. (2014). Forest deforestation dynamics and drivers in Latin America: a review since 1990. *Colombia Forestal*, *17*(2), 233–246. http://doi.org/10.14483/udistrital.jour.colomb.for.2014.2.a07

Armesto, J. J. (2009). Annual reports to council Ecological Society of America August 2009. *Bulletin of the Ecological Society of America*, *90*(4), 360–431. http://doi.org/10.1890/0012-9623-90.4.360

Armesto, J. J., Rozzi, R., Smith-Ramirez, C., & Arroyo, M. T. K. (1998). Conservation targets in South American temperate forests. *Science*, *282*(5392), 1271–1272. http://doi.org/10.1126/science.282.5392.1271

Armesto, J. J., Villagrán, C., & Arroyo, M. K. (1996). *Ecología de los Bosques Nativos de Chile*. Santiago, Chile: Editorial Universitaria.

Aronson, M. F. J., La Sorte, F. A., Nilon, C. H., Katti, M., Goddard, M. A., Lepczyk, C. A., Warren, P. S., Williams, N. S. G., Cilliers, S., Clarkson, B., Dobbs, C., Dolan, R., Hedblom, M., Klotz, S., Kooijmans, J. L., Kühn, I., MacGregor-Fors, I., McDonnell, M. J., Mörtberg, U., Pyšek, P., Siebert, S., Sushinsky, J. R., Werner, P., & Winter, M. (2014). A global analysis of the impacts of urbanization on bird and plant diversity reveals key anthropogenic drivers. *Proceedings of the Royal Society B: Biological Sciences*, *281*(1780). https://doi.org/10.1098/rspb.2013.3330

Arredondo Moreno, T., & Huber-Sannwald, E. (2011). Impacts of drought on agriculture in Northern Mexico. In H. G. Brauch, U. Oswald Spring, C. Mesjasz, J. Grin, P. Kameri-Mboute, B. Chourou, … J. Birkmann (Eds.), *Coping with Global Environmental Change, Disasters and Security* (pp. 875–891). Berlin, Heidelberg: Springer-Verlag. Retrieved from http://link.springer.com/chapter/10.1007/978-3-642-17776-7_51

Arriaga, L. (2009). Implicaciones del cambio de uso de suelo en la biodiversidad de los matorrales xerófilos: un enfoque multiescalar. *Investigación Ambiental: Ciencia Y Política Pública*, *1*(1), 6–16.

Arriaga, L., Castellanos, A. E., Moreno, E., & Alarcón, J. (2004). Potential ecological distribution of alien invasive species and risk assessment: a case study of buffel grass in arid regions of Mexico. *Conservation Biology*, *18*(6), 1504–1514. http://doi.org/10.1111/j.1523-1739.2004.00166.x

Arroyo, M. T. K., & Cavieres, L. (1997). The Mediterranean-type climate flora of Central Chile - What do we know and how can we assure its protection. *Noticiero de Biología*, *5*(2), 48–56.

Arroyo, M. T. K., & Cavieres, L. A. (2013). High-elevation Andean ecosystems. In S. Levin (Ed.), *Encyclopedia of Biodiversity* (pp. 96–110). New Jersey, USA: Elsevier Science and Technology.

Arroyo, M. T. K., Maricorena, C., Matthei, O., & Cavieres, L. (2000). Plant invasions in Chile: present patterns and future predictions. In H. A. Mooney & R. Hobbs (Ed.), *Invasive species in a changing world* (pp. 385–421). New York, USA: Island Press.

Arroyo, M. T. K., Marticorena, C., Matthei, O., Muñoz, M., & Pliscoff, P. (2002). Analysis of the contribution and efficiency of the Santuario de la Naturaleza Yerba Loca, 33° S in protecting the regional vascular plant flora (Metropolitan and Fifth regions of Chile). *Revista Chilena de Historia Natural*, *75*(4), 767–792. http://doi.org/10.4067/S0716-078X2002000400012

Arroyo, M. T. K., Primack, R., & Armesto, J. (1982). Community studies in pollination ecology in the high temperate Andes of central Chile. I. Pollination mechanisms and altitudinal variation. *American Journal of Botany*, *69*(1), 82–97. http://doi.org/10.2307/2442833

Arroyo, M. T. K., Riveros, M., Penaloza, A., Cavieres, L., & Faggi, A. M. (1996). Phytogeographic relationships and regional richness patterns of the cool temperate rainforest flora of southern South America. In R. G. Lawford, E. Fuentes, & P. B. Alaback (Eds.), *High-Latitude Rainforests and Associated Ecosystems of the West Coast of the Americas.* (pp. 134–172). New York, USA: Springer.

Asner, G. P., & Archer, S. R. (2010). Livestock and the global carbon cycle. In H. Steinfeld, H. A. Mooney, F. Schneider, & L. E. Neville (Eds.), *Livestock in a Changing Landscape: Volume 1, Drivers, Consequences, and Responses* (pp. 69–82). Washington, DC., USA: Island Press.

Asner, G. P., Rudel, T. K., Aide, T. M., Defries, R., & Emerson, R. (2009). A contemporary assessment of change in humid tropical forests. *Conservation Biology*, *23*(6), 1386–1395. http://doi.org/10.1111/j.1523-1739.2009.01333.x

Association for Temperate Agroforestry (AFTA). (2017). What is Agroforestry? Retrieved from www.aftaweb.org/about/what-is-agroforestry.html

Astorga, A., Fernández, M., Boschi, E. E., & Lagos, N. (2003). Two oceans, two taxa and one mode of development: latitudinal diversity patterns of South American crabs and test for possible causal processes. *Ecology Letters*, *6*(5), 420–427. http://doi.org/10.1046/j.1461-0248.2003.00445.x

Ataroff, M., & Rada, F. (2000). Deforestation impact on water dynamics in a Venezuelan Andean cloud forest. *Ambio*, *29*(7), 440–444. http://doi.org/10.1579/0044-7447-29.7.440

Auerbach, N. A., Walker, M. D., & Walker, D. A. (1997). Effects of roadside disturbance on substrate and vegetation properties in arctic tundra. *Ecological Applications*, *7*(1), 218–235. http://doi.org/10.1890/1051-0761(1997)007%5B0218:EORDOS%5D2.0.CO;2

Austin, A. T., Piñeiro, G., & Gonzalez-Polo, M. (2006). More is less: Agricultural impacts on the N cycle in Argentina. *Biogeochemistry*, *79*, 45–60. http://doi.org/10.1007/s10533-006-9002-1

Bacheler, N. M., Neal, J. W., & Noble, R. L. (2004). Diet overlap between native bigmouth sleepers (*Gobiomorus dormitor*) and introduced predatory fishes in a Puerto Rico reservoir. *Ecology of Freshwater Fish*, *13*(2), 111–118. http://doi.org/10.1111/j.1600-0633.2004.00040.x

Baena, M. L., & Halfflter, G. (2008). Extinción de especies. In J. Sarukhán (Ed.), *Capital Natural de México, Vol. I: Conocimiento Actual de la Biodiversidad* (pp. 263–282). Mexico City, Mexico: CONABIO.

Baldwin, B. G. (2014). Origins of plant diversity in the California Floristic Province. *Annual Review of Ecology, Evolution, and Systematics*, *45*(1), 347–369. http://doi.org/10.1146/annurev-ecolsys-110512-135847

Baldwin, B. G., Goldman, D. H., Keil, D. J., Patterson, R., Rosatti, T. J., & Wilken, D. H. (Eds.). (2012). *The Jepson Manual: Vascular Plants of California* (2nd ed.). Berkeley, USA: University California Press.

Ballinger, R. E., & Congdon, J. D. (1996). Status of the bunch grass lizard, *Sceloporus scalaris*, in the Chiricahua mountains of southeastern Arizona. *Bulletin of the Maryland Herpetological Society*, *32*, 67–69.

Balvanera, P., Pfisterer, A. B., Buchmann, N., He, J.-S., Nakashizuka, T., Raffaelli, D., & Schmid, B. (2006). Quantifying the evidence for biodiversity effects on ecosystem functioning and services. *Ecology Letters*, *9*(10), 1146–1156. http://doi.org/10.1111/j.1461-0248.2006.00963.x

Balvanera, P., Siddique, I., Dee, L., Paquette, A., Isbell, F., Gonzalez, A., Byrnes, J., O'Connor, M. I., Hungate, B. A., & Griffin, J. N. (2014). Linking biodiversity and ecosystem services: Current uncertainties and the necessary next steps. *BioScience*, *64*, 49–57. https://doi.org/10.1093/biosci/bit003

Banda-R, K., Delgado-Salinas, A., Dexter, K. G., Linares-Palomino, R., Oliveira-Filho, A., Prado, D., Pullan, M., Quintana, C., Riina, R., Rodriguez M., G. M., Weintritt, J., Acevedo-Rodriguez, P., Adarve, J., Alvarez, E., Aranguren B., A., Arteaga, J. C., Aymard, G., Castano, A., Ceballos-Mago, N., Cogollo, A., Cuadros, H., Delgado, F., Devia, W., Duenas, H., Fajardo, L., Fernandez, A., Fernandez, M. A., Franklin, J., Freid, E. H., Galetti, L. A., Gonto, R., Gonzalez-M., R., Graveson, R., Helmer, E. H., Idarraga, A., Lopez, R., Marcano-Vega, H., Martinez, O. G., Maturo, H. M., McDonald, M., McLaren, K., Melo, O., Mijares, F., Mogni, V., Molina, D., Moreno, N. d. P., Nassar, J. M., Neves, D. M., Oakley, L. J., Oatham, M., Olvera-Luna, A. R., Pezzini, F. F., Dominguez, O. J. R., Rios, M. E., Rivera, O., Rodriguez, N., Rojas, A., Sarkinen, T., Sanchez, R., Smith, M., Vargas, C., Villanueva, B., & Pennington, R. T. (2016). Plant diversity patterns in neotropical dry forests and their conservation implications. *Science*, *353*(6306), 1383–1387. https://doi.org/10.1126/science.aaf5080

Barnes, M. (2015). Aichi targets: Protect biodiversity, not just area. *Nature*, *526*(7572), 195–195. http://doi.org/10.1038/526195e

Barreto, R., Ferretti, F., Flemming, J. M., Amorim, A., Andrade, H., Worm, B., & Lessa, R. (2016). Trends in the exploitation of South Atlantic shark populations. *Conservation Biology*, *30*(4), 792–804. http://doi.org/10.1111/cobi.12663

Barrett, K., & Guyer, C. (2008). Differential responses of amphibians and reptiles in riparian and stream habitats to land use disturbances in western Georgia, USA. *Biological Conservation*, *141*(9), 2290–2300. http://doi.org/10.1016/j.biocon.2008.06.019

Barrett, K., Helms, B. S., Guyer, C., & Schoonover, J. E. (2010). Linking process to pattern: Causes of stream-breeding amphibian decline in urbanized watersheds. *Biological Conservation*, *143*, 1998–2005. http://doi.org/10.1016/j.biocon.2010.05.001

Barros, A., & Pickering, C. M. (2014). Non-native plant invasion in relation to tourism use of Aconcagua Park, Argentina, the highest protected area in the Southern Hemisphere. *Mountain Research and Development*, *34*(1), 13–26. http://doi.org/10.1659/MRD-JOURNAL-D-13-00054.1

Bartolomé, M. A. (2006). *Procesos Interculturales. Antropología Política del Pluralismo Cultural en América Latina*. México, D.F., México: Siglo XXI.

Baruch, Z., & Nozawa, S. (2014). Abandoned coffee plantations: biodiversity conservation or path for non-native species ? Case study in a neotropical montane forest. *Interciencia*, *39*(8), 554–561. Retrieved from https://search.proquest.com/docview/1564773458?accountid=14621

Bawa, K. S., Kress, W. J., Nadkarni, N. M., Lele, S., Raven, P. H., Janzen, D. H., Lugo, A. E., Ashton, P. S., & Lovejoy, T. E. (2004). Tropical ecosystems into the 21st century. *Science*, *306*, 227–228. https://doi.org/10.1126/science.306.5694.227b

Beckage, B., Osborne, B., Gavin, D. G., Pucko, C., Siccama, T., & Perkins, T. (2008). A rapid upward shift of a forest ecotone during 40 years of warming in the Green Mountains of Vermont. *PNAS*, *105*, 4197–4202. http://doi.org/10.1073/pnas.0708921105

Beddow, J. M., Pardey, P. G., Koo, J., & Wood, S. (2010). The changing landscape of global agriculture. In J. M. Alston, B. A. Babcock, & P. G. Pardey (Eds.), *The Shifting Patterns of Agricultural Production and Productivity Worldwide*. Ames, USA: Iowa State University, The Midwest Agribusiness Trade Research and Information Center (MATRIC).

Beever, E. A., Perrine, J. D., Rickman, T., Flores, M., Clark, J. P., Waters, C., ., & Goehring, K. E. (2016). Pika (Ochotona princeps) losses from two isolated regions reflect temperature and water balance, but reflect habitat area in a mainland region. *Journal of Mammalogy*, *97*(6), 1495–1511. http://doi.org/10.1093/jmammal/gyw128

Beever, E. A., Ray, C., Wilkening, J. L., Brussard, P. F., & Mote, P. W. (2011). Contemporary climate change alters the pace and drivers of extinction. *Global Change Biology*, *17*(6), 2054–2070. http://doi.org/10.1111/j.1365-2486.2010.02389.x

Bellard, C., Cassey, P., & Blackburn, T. M. (2016). Alien species as a driver of recent extinctions. *Biology Letters*, *12*(2), 20150623. http://doi.org/10.1098/rsbl.2015.0623

Bello, C., Galetti, M., Pizo, M. A., Magnago, L. F. S., Rocha, M. F., Lima, R. A. F., Peres, C. A., Ovaskainen, O., & Jordano, P. (2015). Defaunation affects carbon storage in tropical forests. *Science Advances*, *1*(11), e1501105. https://doi.org/10.1126/sciadv.1501105

Bello, J. C., Báez, M., Gomez, M. F., Orrego, O., & Nägele, L. (Eds.). (2014). *Biodiversidad 2014. Estado y Tendencias de la Biodiversidad Continental de Colombia*. Bogotá, D.C., Colombia: Instituto Alexander von Humboldt. Retrieved from http://www.humboldt.org.co/es/estado-de-los-recursos-naturales/item/898-bio2015

Belnap, J., & Eldridge, D. (2003). Disturbance and recovery of biological soil crusts. In J. Belnap & O. L. Lange (Eds.), *Biological Soil Crusts: Structure, Function, and Management* (pp. 363–383). Berlin, Germany: Springer-Verlag.

Belnap, J., Weber, B., & Büdel, B. (2016). Biological soil crusts as an organizing principle in drylands. In J. Weber, B, Büdel, B, Belnap (Ed.), *Biological Soil Crusts: An Organizing Principle in Drylands* (pp. 3–13). Switzerland: Springer International Publishing. Retrieved from http://link.springer.com/10.1007/978-3-319-30214-0_1

Benayas, R. J. M., Newton, A. C., Diaz, A., & Bullock, J. M. (2009). Enhancement of biodiversity and ecosystem services by ecological restoration: a meta-analysis. *Science*, *325*(5944), 1121–1124. http://doi.org/10.1126/science.1172460

Beneyto, D., Monros, J. S., & Piculo, R. (2013). The Raffia-swamps as sources or sinks of avifauna: a first approach to the problem. *Revista de Biologia Tropical*, *61*(supl. 1), 131–142. http://doi.org/10.15517/rbt.v61i1.23184

Beninde, J., Veith, M., & Hochkirch, A. (2015). Biodiversity in cities needs space: a meta-analysis of factors determining intra-urban biodiversity variation. *Ecology Letters*, *18*, 581–592. http://doi.org/10.1111/ele.12427

Bennett, E., & Thorpe, A. (2008). Review of river fisheries valuation in Central and South America. In A. Neiland & C. Béné (Eds.), *Tropical River Fisheries Valuation: Background Papers to a Global Synthesis* (pp. 3–44). Penang, Malaysia: The WorldFish Center Studies and Reviews.

Benson, A. J. (2012). The Exotic Zebra Mussel. U.S. Fish and Wildlife Service. Retrieved from http://www.fws.gov/midwest/endangered/clams/zebra.html.

Benson, A. J., & Boydstun, C. P. (1999). Documenting over a century of aquatic introductions in the United States. In R. Claudi & J. H. Leach (Eds.), *Nonindigenous Freshwater Organisms: Vectors, Biology, and Impacts* (pp. 1–31). Boca Raton, USA: Lewis Publishers.

Benson, A. J., Marelli, D. C., Frischer, M. E., Danforth, J. M., & Williams, J. D. (2001). Establishment of the green mussel *Perna viridis* (Linnaeus 1758), (Mollusca: Mytilidae) on the west coast of Florida. *Journal of Shellfish Research*, *20*(1), 21–29.

Benson, J. F., Mahoney, P. J., Sikich, J. A., Serieys, L. E. K., Pollinger, J. P., Ernest, H. B., & Riley, S. P. (2016). Interactions between demography, genetics, and landscape connectivity increase extinction probability for a small population of large carnivores in a major metropolitan area. *Proceedings of the Royal Society B: Biological Sciences*, *283*(1837), 20160957. http://doi.org/10.1098/rspb.2016.0957

Berkes, F. (2008). *Sacred Ecology*. New York, USA: Routledge.

Berkes, F., & Jolly, D. (2002). Adapting to climate change: Social-ecological resilience in a Canadian western arctic community. *Conservation Ecology*, *5*(2), 18. Retrieved from http://www.consecol.org/vol5/iss2/art18/

Berthrong, S. T., Piñeiro, G., Jobbágy, E. G., & Jackson, R. B. (2012). Soil C and N changes with afforestation of grasslands across gradients of precipitation and plantation age. *Ecological Applications*, *22*(1), 76–86. http://doi.org/10.1890/10-2210.1

Bertolin, M. L., Urretavizcaya, M. F., & Defosse, G. E. (2015). Fire emissions and carbon uptake in severely burned lenga beech (Nothofagus pumilio) forests of Patagonia, Argentina. *Fire Ecology*, *11*(1), 32–54. http://doi.org/10.4996/fireecology.1101032

Beuchle, R., Grecchi, R. C., Shimabukuro, Y. E., Seliger, R., Eva, H. D., Sano, E., & Achard, F. (2015). Land cover changes in the Brazilian Cerrado and Caatinga biomes from 1990 to 2010 based on a systematic remote sensing sampling approach. *Applied Geography*, *58*, 116–127. http://doi.org/10.1016/j.apgeog.2015.01.017

Bhatt, U. S., Walker, D. A., Raynolds, M. K., Comiso, J. C., Epstein, H. E., Jia, G., Gens, R., Pinzon, J. E., Tucker, C. J., Tweedie, C. E., & Webber, P. J. (2010). Circumpolar Arctic tundra vegetation change is linked to sea ice decline. *Earth Interactions*, *14*(8), 1–20. https://doi.org/10.1175/2010EI315.1

Bianchi, C. A., & Haig, S. M. (2013). Deforestation trends of tropical dry forests in Central Brazil. *Biotropica*, *45*(3), 395–400. http://doi.org/10.1111/btp.12010

Bilenca, D., & Miñarro, F. (2004). *Identificación de Áreas Valiosas de Pastizal (AVPs) en las Pampas y Campos* *de Argentina, Uruguay y Sur de Brazil*. Buenos Aires, Argentina: Fundación Vida Silvestre.

BirdLife International. (2015). DataZone. Retrieved from http://www.birdlife.org/datazone/site/search

BirdLife International, & NatureServe. (2012). Bird species distribution maps of the world. Version 2.0. Cambridge, UK and Arlington, USA.

Birx-Raybuck, D. A., Price, S. J., & Dorcas, M. E. (2010). Pond age and riparian zone proximity influence anuran occupancy of urban retention ponds. *Urban Ecosystems*, *13*(2), 181–190. http://doi.org/10.1007/s11252-009-0116-9

Blair, R. B., & Launer, A. E. (1997). Butterfly diversity and human land use: Species assemblages along an urban gradient. *Biological Conservation*, *80*(1), 113–125. http://doi.org/10.1016/S0006-3207(96)00056-0

Bogan, A. E. (2008). Global diversity of freshwater mussels (Mollusca, Bivalvia) in freshwater. *Hydrobiologia*, *595*(1), 139–147. http://doi.org/10.1007/s10750-007-9011-7

Bonilla-Murillo, F., Beneyto, D., & Sasa, M. (2013). Amphibians and reptiles in the swamps dominated by the palm *Raphia taedigera* (Arecaceae) in northeastern Costa Rica. *Revista de Biologia Tropical*, *61*(Suppl. 1), 143–161. http://doi.org/10.15517/rbt.v61i1.23185

Bonoff, M. B., & Janzen, D. H. (1980). Small terrestrial rodents in eleven habitats in Santa Rosa National Park. Costa Rica. *Brenesia*, *17*, 163–174.

Borgnia, M., Vila, B. L., & Cassini, M. H. (2008). Interaction between wild camelids and livestock in an Andean semi-desert. *Journal of Arid Environments*, *72*(12), 2150–2158. http://doi.org/10.1016/j.jaridenv.2008.07.012

Bortolus, A., Schwindt, E., Bouza, P. J., & Idaszkin, Y. L. (2009). A characterization of Patagonian salt marshes. *Wetlands*, *29*(2), 772–780. http://doi.org/10.1672/07-195.1

Bosch, J. M., & Hewlett, J. D. (1982). A review of catchment experiments to determine the effect of vegetation changes on water yield and evapotranspiration. *Journal of Hydrology*, *55*(1–4), 3–23. http://doi.org/10.1016/0022-1694(82)90117-2

Bouchenak-Khelladi, Y., Muasya, A. M., & Linder, H. P. (2014). A revised evolutionary history of Poales: origins and diversification. *Botanical Journal of the Linnean Society*, *175*(1), 4–16. http://doi.org/10.1111/boj.12160

Bour, R. (2008). Global diversity of turtles (Chelonii; Reptilia) in freshwater. *Hydrobiologia*, *595*(1), 593–598. http://doi.org/10.1007/s10750-007-9244-5

Bowman, W. D., & Seastedt, T. R. (Eds.). (2001). *Structure and Function of an Alpine Ecosystem: Niwot Ridge, Colorado*. New York, USA: Oxford University Press.

Bozec, Y.-M., O'Farrell, S., Bruggemann, J. H., Luckhurst, B. E., & Mumby, P. J. (2016). Tradeoffs between fisheries harvest and the resilience of coral reefs. *PNAS*, *113*(16), 4536–4541. http://doi.org/10.1073/pnas.1601529113

Bradley, R. D., Ammerma, L. K., Baker, R. J., Bradley, L. C., Cook, J. A., Dowler, R. C., Jones, C., Schmidley, D. J., Stangl Jr, R., Van Den Bussche, R., & Würsig, B. (2014). Revised checklist of North American mammals North of Mexico. *Occasional Papers Museum of Texas Tech University*, *327*, 1–27.

Bradshaw, C. J. A., Leroy, B., Bellard, C., Roiz, D., Albert, C., Fournier, A., Barbet-Massin, M., Salles, J.-M., Simard, F., & Courchamp, F. (2016). Massive yet grossly underestimated global costs of invasive insects. *Nature Communications*, *7*, 12986. https://doi.org/10.1038/ncomms12986

Brandt, R., Mathez-Stiefel, S.-L., Lachmuth, S., Hensen, I., & Rist, S. (2013). Knowledge and valuation of Andean agroforestry species: the role of sex, age, and migration among members of a rural community in Bolivia. *Journal of Ethnobiology and Ethnomedicine*, *9*, 83. http://doi.org/10.1186/1746-4269-9-83

Brauman, K. A., Daily, G. C., Duarte, T. K., & Mooney, H. A. (2007). The nature and value of ecosystem services: An overview highlighting hydrologic services. *Annual Review of Environment and Resources*, *32*(1), 67–98. http://doi.org/10.1146/annurev.energy.32.031306.102758

Braun, A. C., & Koch, B. (2016). Estimating impacts of plantation forestry on plant biodiversity in southern Chile - a spatially explicit modelling approach. *Environmental Monitoring and Assessment*, *188*(10), 564. http://doi.org/10.1007/s10661-016-5547-1

Brenner, M., Schelske, C. L., & Keenan, L. W. (2001). Historical rates of sediment and nutrient accumulation in marshes of the Upper St. Johns River Basin, Florida. *Journal of Paleolimnology*, *26*(3), 241–257.

http://doi.org/10.1023/A:1017578330641
Brick Peres, M., Barreto, R., Lessa, R., Vooren, C., Charvet, P., & Rosa, R. (2012). Heavy fishing puts Brazilian sharks and rays in great trouble. In *6th World Fisheries Congress, Sustainable Fisheries in a Changing World*. Edinburgh, Scotland.
Bridi, R., & Montenegro, G. (2017). The value of chilean honey: Floral origin related to their antioxidant and antibacterial activities. In V. A. A. Toledo (Ed.), *Honey Analysis* (pp. 63–78). InTech, Open Access. Retrieved from https://www.intechopen.com/books/honey-analysis/the-value-of-chilean-honey-floral-origin-related-to-their-antioxidant-and-antibacterial-activities
Briggs, C. J., Vredenburg, V. T., Knapp, R. A., & Rachowicz, L. J. (2005). Investigating the population-level effects of Chytridiomycosis: An emerging infectious disease of amphibieans. *Ecology*, *86*, 3149–3159. http://doi.org/10.1890/04-1428
Bring, A., Fedorova, I., Dibike, Y., Hinzman, L., Mard, J., Mernild, S. H., Prowse, T., Semenova, O., Stuefer, S. L., & Woo, M. K. (2016). Arctic terrestrial hydrology: A synthesis of processes, regional effects, and research challenges. *Journal of Geophysical Research*, *121*(3), 621–649.
Brinson, M. M., & Malvarez, A. I. (2002). Temperate freshwater wetlands: types, status, and threats. *Environmental Conservation*, *29*(2), 115–133. http://doi.org/10.1017/S0376892902000085
Bromberg, K. D., & Bertness, M. D. (2005). Reconstructing New England salt marsh losses using historical maps. *Estuaries*, *28*(6), 823–832. http://doi.org/10.1007/BF02696012
Brooker, R. W., Maestre, F. T., Callaway, R. M., Lortie, C. L., Cavieres, L. A., Kunstler, G., Liancourt, P., Tielbörger, K., Travis, J. M. J., Anthelme, F., Armas, C., Coll, L., Corcket, E., Delzon, S., Forey, E., Kikvidze, Z., Olofsson, J., Pugnaire, F., Quiroz, C. L., Saccone, P., Schiffers, K., Seifan, M., Touzard, B., & Michalet, R. (2008). Facilitation in plant communities: the past, the present, and the future. *Journal of Ecology*, *96*(1), 18–34. https://doi.org/10.1111/j.1365-2745.2007.01295.x
Brooks, M. (2009). Spatial and temporal distribution of nonnative plants in upland areas of the Mojave Desert. In R. Webb, L. Fenstermaker, J. Heaton, D. Hughson, E. McDonald, & D. Miller (Eds.), *The Mojave Desert: Ecosystem Processes and Sustainability* (pp. 101–124). Reno, USA: University of Nevada Press.
Brooks, T. M., Akçakaya, H. R., Burgess, N. D., Butchart, S. H. M., Hilton-Taylor, C., Hoffmann, M., Juffe-Bignoli, D., Kingston, N., MacSharry, B., Parr, M., Perianin, L., Regan, E. C., Rodrigues, A. S. L., Rondinini, C., Shennan-Farpon, Y., & Young, B. E. (2016). Analysing biodiversity and conservation knowledge products to support regional environmental assessments. *Scientific Data*, *3*. https://doi.org/10.1038/sdata.2016.7
Brown, A. D. (1990). Epiphytism in the montane forests of El Rey National Park in Argentina: Floristic composition and distribution pattern. *Revista de Biología Tropical*, *38*(2A), 155–166.
Brown, A. E., Zhang, L., McMahon, T. A., Western, A. W., & Vertessy, R. A. (2005). A review of paired catchment studies for determining changes in water yield resulting from alterations in vegetation. *Journal of Hydrology*, *310*(1–4), 28–61. http://doi.org/10.1016/j.jhydrol.2004.12.010
Brown, M. T., Cohen, M. J., Bardi, E., & Ingwersen, W. W. (2006). Species diversity in the Florida Everglades, USA: A systems approach to calculating biodiversity. *Aquatic Sciences*, *68*, 254–277. http://doi.org/10.1007/s00027-006-0854-1
Brown, S. J., & Maceina, M. J. (2002). The influence of disparate levels of submersed aquatic vegetation on largemouth bass population characteristics in a Georgia reservoir. *Journal of Aquatic Plant Management*, *40*, 28–35.
Bucher, E. (1982). Chaco and Caatinga - South American arid savannas, woodlands and thickets. In B. I. Huntley & B. H. Walker (Eds.), *Ecology of Tropical Savannas* (pp. 48–79). Berlin, Germany: Springer-Verlag.
Bucher, E. H., Echevarria, A. L., Juri, M. D., & Chani, J. M. (2000). Long-term survey of Chilean flamingo breeding colonies on Mar Chiquita lake, Córdoba, Argentina. *Waterbirds: The International Journal of Waterbird Biology*, *23*, 114–118. http://doi.org/10.2307/1522155
Buckley, L. B., & Jetz, W. (2007). Environmental and historical constraints on global patterns of amphibian richness. *Proceedings of the Royal Society B: Biological Sciences*, *274*(1614), 1167–1173. http://doi.org/10.1098/rspb.2006.0436
Buddo, D. S. A., Steele, R. D., & D'Oyen, E. R. (2003). Distribution of the invasive Indo-Pacific green mussel,

Perna viridis, in Kingston Harbour, Jamaica. *Bulletin of Marine Science*, *73*(2), 433–441.
Bullock, J. M., Pywell, R. F., & Walker, K. J. (2007). Long-term enhancement of agricultural production by restoration of biodiversity. *Journal of Applied Ecology*, *44*(1), 6–12. http://doi.org/10.1111/j.1365-2664.2006.01252.x
Burge, D. O., Thorne, J. H., Harrison, S. P., O'Brien, B. C., Rebman, J. P., Shevock, J. R., Alverson, E. R., Hardison, L. K., RodrÍguez, J. D., Junak, S. A., Oberbauer, T. A., Riemann, H., Vanderplank, S. E., & Barry, T. (2016). Plant diversity and endemism in the California Floristic Province. *Madroño*, *63*(2), 3–206. https://doi.org/10.3120/madr-63-02-3-206.1
Burkhead, N. M. (2012). Extinction rates in North American freshwater fishes, 1900–2010. *BioScience*, *62*, 798–808. http://doi.org/10.1525/bio.2012.62.9.5
Burridge, L., Weis, J. S., Cabello, F., Pizarro, J., & Bostick, K. (2010). Chemical use in salmon aquaculture: A review of current practices and possible environmental effects. *Aquaculture*, *306*(1–4), 7–23. http://doi.org/10.1016/j.aquaculture.2010.05.020
Buschmann, A. H., Riquelme, V. A., Hernández-González, M. C., Varela, D., Jiménez, J. E., Henríquez, L. A., Vergara, P. A., Guíñez, R., & Filún, L. (2006). A review of the impacts of salmonid farming on marine coastal ecosystems in the southeast Pacific. *ICES Journal of Marine Science*, *63*(7), 1338–1345. https://doi.org/10.1016/j.icesjms.2006.04.021
Bustamante, R. O., & Castor, C. (1998). The decline of an endangered temperate ecosystem: the ruil (*Nothofagus alessandrii*) forest in central Chile. *Biodiversity and Conservation*, *7*(12), 1607–1626. http://doi.org/10.1023/A:1008856912888
Butchart, S. H. M., Scharlemann, J. P. W., Evans, M. I., Quader, S., Aricò, S., Arinaitwe, J., Balman, M., Bennun, L. A., Bertzky, B., Besançon, C., Boucher, T. M., Brooks, T. M., Burfield, I. J., Burgess, N. D., Chan, S., Clay, R. P., Crosby, M. J., Davidson, N. C., de Silva, N., Devenish, C., Dutson, G. C. L., Fernández, D. F. D., Fishpool, L. D. C., Fitzgerald, C., Foster, M., Heath, M. F., Hockings, M., Hoffmann, M., Knox, D., Larsen, F. W., Lamoreux, J. F., Loucks, C., May, I., Millett, J., Molloy, D., Morling, P., Parr, M., Ricketts, T. H., Seddon, N., Skolnik, B., Stuart, S. N., Upgren, A., & Woodley, S. (2012). Protecting important sites for biodiversity contributes to meeting global conservation targets. *PLoS ONE*, *7*(3), e32529. https://doi.org/10.1371/journal.pone.0032529
Buytaert, W., Cuesta-Camacho, F., & Tobón, C. (2011). Potential impacts of climate change on the environmental services of humid tropical alpine regions. *Global Ecology and Biogeography*, *20*, 19–33. http://doi.org/10.1111/j.1466-8238.2010.00585.x
Buytaert, W., Wyseure, G., De Bievre, B., & Deckers, J. (2005). The effect of land-use changes on the hydrological behaviour of Histic Andosols in south Ecuador. *Hydrological Processes*, *19*(20), 3985–3997. http://doi.org/10.1002/hyp.5867
Caballero, J., & Cortés, L. (2001). Percepción, uso y manejo tradicional de los recursos vegetales en México. In B. Rendón Aguilar, S. Rebollar Domínguez, J. Caballero Nieto, & M. A. Martínez Alfaro (Eds.), *Plantas, Cultura y Sociedad* (pp. 79–100). México D.F., México: Universidad Autónoma Metropolitana y Secretaría del Medio Ambiente, Recursos Naturales y Pesca.
Cabette, H. S. R., Souza, J. R., Shimano, Y., & Juen, L. (2017). Effects of changes in the riparian forest on the butterfly community (Insecta: Lepidoptera) in Cerrado areas. *Revista Brasileira de Entomologia*, *61*(1), 43–50. http://doi.org/10.1016/j.rbe.2016.10.004
Cáceres, N. C., Nápoli, R. P., Casella, J., & Hannibal, W. (2010). Mammals in a fragmented savannah landscape in south-western Brazil. *Journal of Natural History*, *44*(7–8), 491–512. http://doi.org/10.1080/00222930903477768
Cadena-Vargas, C. E., & Sarmiento, C. E. (2016). Cambios en las coberturas paramunas. Las Amenazas de los páramos de Colombia. In M. F. Gómez, L. A. Moreno, G. I. Andrade, & C. Rueda (Eds.), *Biodiversidad 2015. Estado y Tendencias de la Biodiversidad Continental de Colombia*. Bogotá, D.C., Colombia: Instituto Alexander von Humboldt.
Califano, L. M., & Echazú, F. (2013). Etnobotánica en comunidades pastoriles: Conocimiento tradicional sobre especies tóxicas para el ganado en la cuenca del río Iruya (Salta, Argentina). *Boletín de La Sociedad Argentina de Botánica*, *48*(2), 365–375.
California Invasive Plant Council. (2017). California Invasive Plant Council. Retrieved from http://cal-ipc.org/
California Natural Resources Agency. (2015). *California Department of Fish and Wildlife*. Retrieved from

https://www.wildlife.ca.gov/Conservation/Plants/Invasives

Callaghan, T. V., Björn, L. O., Chernov, Y., Chapin, T., Christensen, T. R., Huntley, B., … Zöckler, C. (2004). Biodiversity, distributions and adaptations of Arctic species in the context of environmental change. *Ambio*, *33*(7), 404–417. http://doi.org/10.1579/0044-7447-33.7.404

Calvo-Alvarado, J., McLennan, B., Sánchez-Azofeifa, A., & Garvin, T. (2009). Deforestation and forest restoration in Guanacaste, Costa Rica: Putting conservation policies in context. *Forest Ecology and Management*, *258*(6), 931–940. http://doi.org/10.1016/j.foreco.2008.10.035

Calvo-Alvarado, J., Sánchez-Azofeifa, A., & Portillo-Quintero, C. (2013). Neotropical seasonally dry forests. In S. Levin (Ed.), *Encyclopedia of Biodiversity* (pp. 488–500). Amsterdam, The Netherlands: Academic Press. Retrieved from http://linkinghub.elsevier.com/retrieve/pii/B9780123847195003543

Cameron, R. W. ., Blanuša, T., Taylor, J. E., Salisbury, A., Halstead, A. J., Henricot, B., & Thompson, K. (2012). The domestic garden – Its contribution to urban green infrastructure. *Urban Forestry & Urban Greening*, *11*(2), 129–137. http://doi.org/10.1016/j.ufug.2012.01.002

Canfield, D. E., Glazer, A. N., & Falkowski, P. G. (2010). The evolution and future of Earth's nitrogen cycle. *Science*, *330*(6001), 192–196. http://doi.org/10.1126/science.1186120

Cano-Ortiz, A., Musarella, C. M., Piñar-Fuentes, J. C., Pinto-Gomes, C., & Cano-Carmona, E. (2015). Forests and landscapes of Dominican Republic. *British Journal of Applied Science & Technology*, *9*(3), 231–242. http://doi.org/10.9734/BJAST/2015/17507

Carabias, J., Sarukhán, J., De la Maza, J., & Galindo, C. (2010). *Patrimonio Natural de México. Cien Casos de Éxito.* Mexico City, Mexico: México, Comisión Nacional para el Conocimiento y Uso de la Biodiversidad. Retrieved from http://www.biodiversidad.gob.mx/pais/cien_casos/pdf/Cien casos.pdf

PLoS ONE, *7*(3), e32529. https://doi.org/10.1371/journal.pone.0032529

Cardinale, B. J., Duffy, J. E., Gonzalez, A., Hooper, D. U., Perrings, C., Venail, P., Narwani, A., Mace, G. M., Tilman, D., A.Wardle, D., Kinzig, A. P., Daily, G. C., Loreau, M., Grace, J. B., Larigauderie, A., Srivastava, D. S., & Naeem, S. (2012). Erratum: Corrigendum: Biodiversity loss and its impact on humanity. *Nature*, *489*(7415), 326–326. https://doi.org/10.1038/nature11373

Cardinale, B. J., Matulich, K. L., Hooper, D. U., Byrnes, J. E., Duffy, E., Gamfeldt, L., Balvanera, P., O'Connor, M. I., & Gonzalez, A. (2011). The functional role of producer diversity in ecosystems. *American Journal of Botany*, *98*(3), 572–592. https://doi.org/10.3732/ajb.1000364

Cardinale, B. J., Wright, J. P., Cadotte, M. W., Carroll, I. T., Hector, A., Srivastava, D. S., Loreau, M., & Weis, J. J. (2007). Impacts of plant diversity on biomass production increase through time because of species complementarity. *PNAS*, *104*(46), 18123–18128. https://doi.org/10.1073/pnas.0709069104

Cardoso, D., Särkinen, T., Alexander, S., Amorim, A. M., Bittrich, V., Celis, M., Daly, D. C., Fiaschi, P., Funk, V. A., Giacomin, L. L., Goldenberg, R., Heiden, G., Iganci, J., Kelloff, C. L., Knapp, S., Cavalcante de Lima, H., Machado, A. F. P., dos Santos, R. M., Mello-Silva, R., Michelangeli, F. A., Mitchell, J., Moonlight, P., de Moraes, P. L. R., Mori, S. A., Nunes, T. S., Pennington, T. D., Pirani, J. R., Prance, G. T., de Queiroz, L. P., Rapini, A., Riina, R., Rincon, C. A. V., Roque, N., Shimizu, G., Sobral, M., Stehmann, J. R., Stevens, W. D., Taylor, C. M., Trovó, M., van den Berg, C., van der Werff, H., Viana, P. L., Zartman, C. E., & Forzza, R. C. (2017). Amazon plant diversity revealed by a taxonomically verified species list. *PNAS*, *114*(40), 10695–10700. https://doi.org/10.1073/pnas.1706756114

Carlton, J. T., Chapman, J. W., Geller, J. B., Miller, J. A., Carlton, D. A., McCuller, M. I., Treneman, N. C., Steves, B. P., & Ruiz, G. M. (2017). Tsunami-driven rafting: Transoceanic species dispersal and implications for marine biogeography. *Science*, *357*(6358), 1402–1406. https://doi.org/10.1126/science.aao1498

Carmignotto, A. P., De Vivo, M., & Langguth, A. (2012). Mammals of the Cerrado and Caatinga: distribution patterns of the tropical open biomes of central South America. In B. D. Patterson & L. P. Costa (Eds.), *Bones, Clones and Biomes. The History and Geography of Recent Neotropical Mammals* (pp. 307–350). Chicago, USA.: University of Chicago Press.

Carvalho, F. M., De Marco, P., & Ferreira, L. G. (2009). The Cerrado into-pieces: Habitat fragmentation as a function of landscape use in the savannas of central Brazil. *Biological Conservation*, *142*(7), 1392–

1403.

Casas, A., Moreno-Calles, A. I., Vallejo, M., & Parra, F. (2016). Importancia actual y potencial de los recursos genéticos. In A. Casas, J. Torres-Guevara, & F. Parra (Eds.), *Domesticación en el Continente Americano Vol. 1. Investigación Manejo de Biodiversidad y Evolución Dirigida por las Culturas del Nuevo Mundo* (pp. 51–74). México D.F., México and Lima, Perú: Universidad Nacional Autónoma de México/Universidad Nacional Agraria La Molina.

Casas, A., Otero-Arnaiz, A., Perez-Negron, E., & Valiente-Banuet, A. (2007). In situ management and domestication of plants in Mesoamerica. *Annals of Botany*, *100*(5), 1101–1115. http://doi.org/10.1093/aob/mcm126

Casas, A., Parra-Rodinel, F., Rangel-Landa, S., Blancas, J., Vallejo, M., Moreno-Calles, A. I., Guillén, S., Torres-García, I., Delgado-Lemus, A., Pérez-Negrón, E., Figueredo, C. J., Cruse-Sanders, J. M., Farfán-Heredia, B., Solís, L., Aguirre-Dugua, X., Otero-Arnaiz, A., Alvarado-Sizzo, H., & Camou-Guerrero, A. (2017). Manejo y domesticación de plantas en Mesoamérica. Una estrategia de investigación y estado del conocimiento sobre los recursos genéticos en México. In A. Casas, J. Torres-Guevara, & F. Parra-Rondinel (Eds.), *Domesticación en el Continente Americano Vol. 2. Investigación para el Manejo Sustentable de Recursos Genéticos en el Nuevo Mundo* (pp. 69–102). México D.F., México and Lima, Perú: Universidad Nacional Autónoma de México/Universidad Nacional Agraria La Molina/CONACYT.

Casas, A., Parra-Rondinel, F., Torres-García, I., Rangel-Landa, S., Zarazúa, M., & Torres-Guevara, J. (2017). Estudios y patrones continentales de domesticación y manejo de recursos genéticos: perspectivas. In A. Casas, J. Torres-Guevara, & F. Parra-Rondinel (Eds.), *Domesticación en el Continente Americano Vol. 2. Investigación para el Manejo Sustentable de Recursos Genéticos en el Nuevo Mundo* (pp. 537–569). México D.F., México and Lima, Perú: Universidad Nacional Autónoma de México/Universidad Nacional Agraria La Molina/CONACYT.

Casas, A., Parra, F., Blancas, J., Rangel-Landa, S., Vallejo, M., Figueredo, C. J., & Moreno-Calles, A. I. (2016). Origen de la domesticación y la agricultura: cómo y por qué. In A. Casas, J. Torres-Guevara, & F. Parra (Eds.), *Domesticación en el Continente Americano Vol. 1. Investigación Manejo de Biodiversidad y Evolución Dirigida por las Culturas del Nuevo Mundo* (pp. 189–224). México D.F., México and Lima, Perú: Universidad Nacional Autónoma de México/Universidad Nacional Agraria La Molina.

Cashman, A. (2014). Water security and services in the Caribbean. *Water*, *6*(5), 1187–1203.http://doi.org/10.3390/w6051187

Casner, K. L., Forister, M. L., O'Brien, J. M., Thorne, J., Waetjen, D., & Shapiro, A. M. (2014). Contribution of urban expansion and a changing climate to decline of a butterfly fauna. *Conservation Biology*, *28*(3), 773–782. http://doi.org/10.1111/cobi.12241

Castello, L., McGrath, D. G., Hess, L. L., Coe, M. T., Lefebvre, P. A., Petry, P., Macedo, M. N., Renó, V. F., & Arantes, C. C. (2013). The vulnerability of Amazon freshwater ecosystems. *Conservation Letters*, *6*(4), 217–229. https://doi.org/10.1111/conl.12008

Castillo, L. E., Ruepert, C., & Solis, E. (2000). Pesticide residues in the aquatic environment of banana plantation areas in the north Atlantic zone of Costa Rica. *Environmental Toxicology and Chemistry*, *19*, 1942–1950. http://doi.org/10.1002/etc.5620190802

Cavalcanti, M. C. B. T., Ramos, M. A., Araújo, E. L., & Albuquerque, U. P. (2015). Implications from the use of non-timber forest products on the consumption of wood as a fuel source in human-dominated semiarid landscapes. *Environmental Management*, *56*(2), 389–401. http://doi.org/10.1007/s00267-015-0510-4

Cavanaugh, K. C., Kellner, J. R., Forde, A. J., Gruner, D. S., Parker, J. D., Rodriguez, W., & Feller, I. C. (2014). Poleward expansion of mangroves is a threshold response to decreased frequency of extreme cold events. *PNAS*, *111*, 723–727. http://doi.org/10.1073/pnas.1315800111

Cavender-Bares, J., & Reich, P. B. (2012). Shocks to the system: community assembly of the oak savanna in a 40-year fire frequency experiment. *Ecology*, *93*(Special Issue), 52–69. http://doi.org/10.1890/11-0502.1

Cavia, R., Cueto, G. R., & Suárez, O. V. (2009). Changes in rodent communities according to the landscape structure in an urban ecosystem. *Landscape and Urban Planning*, *90*(1–2), 11–19.

http://doi.org/10.1016/j.landurbplan.2008.10.017

Cavicchia, M., & García, G. V. (2012). Riqueza y composición de especies de aves rapaces (Falconiformes y Strigiformes) de la ciudad de Buenos Aires, Argentina. *Hornero*, *27*(2), 159–166.

Caziani, S. M., Derlindati, E. J., Tálamo, A., Sureda, A. L., Trucco, C. E., & Nicolossi, G. (2001). Waterbird richness in Altiplano wetlands of northwestern Argentina. *Waterbirds*, *24*, 103–117. http://doi.org/10.2307/1522249

Ceballos, G., & Ehrlich, P. R. (2009). Discoveries of new mammal species and their implications for conservation and ecosystem services. *PNAS*, *106*(10), 3841–3846. http://doi.org/10.1073/pnas.0812419106

Ceballos, G., Garcia-Aguayo, A., Rodriguez, P., & Noguera, F. (1993). *Plan de Manejo de la Reserva Ecológica de Chamela-Cuixmala*. México City, Mexico: Fundacion Ecologia de Cuixmala.

Cendrero, A., Terán, D. de, Ramón, J., González, D., Mascitti, V., Rotondaro, R., & Tecchi, R. (1993). Environmental diagnosis for planning and management in the high Andean region: The biosphere reserve of Pozuelos, Argentina. *Environmental Management*, *17*(5), 683–703. http://doi.org/10.1007/BF02393729

CEPAL. (2014). *Panorama Social de America Latina (LC/G.2635-P)*. Santiago, Chile.

Chace, J. F., & Walsh, J. J. (2006). Urban effects on native avifauna: A review. *Landscape and Urban Planning*, *74*(1), 46–69. http://doi.org/10.1016/j.landurbplan.2004.08.007

Chacón, M., Pickersgill, S., & Debouck, D. (2005). Domestication patterns in common bean (*Phaseolus vulgaris* L.) and the origin of the Mesoamerican and Andean cultivated races. *Theoretical and Applied Genetics*, *110*(3), 432–444. http://doi.org/10.1007/s00122-004-1842-2

Challenger, A., & Dirzo, R. (2009). Factores de cambio y estado de la biodiversidad. In J. Sarukhán (Ed.), *Capital Natural de México, Vol. II: Estado de Conservación y Tendencias de Cambio* (pp. 37–73). Mexico City, Mexico: CONABIO.

Challenger, A., & Soberón, J. (2008). Los ecosistemas terrestres. In J. Sarukhán (Ed.), *Capital Natural de México, Vol. I: Conocimiento Actual de la Biodiversidad* (pp. 87–108). Sarukhán, José: CONABIO.

Chapin, F. S., Hoel, M., Carpenter, S. R., Lubchenco, J., Walker, B., Callaghan, T. V, … Zimov, S. A. (2006). Building resilience and adaptation to manage Arctic change. *Ambio*, *35*(4), 198–202. http://doi.org/10.1579/0044-7447(2006)35%5B198:braatm%5D2.0.co;2

Chapin, F. S., Matson, P. A., & Vitousek, P. M. (2012). *Principles of Terrestrial Ecosystem Ecology*. New York, USA: Springer-Verlag.

Chapin, F. S., & Shaver, G. R. (1985). Individualistic growth response of tundra plant species to environmental manipulations in the field. *Ecology*, *66*(2), 564–576. http://doi.org/10.2307/1940405

Chapin, F. S., Shaver, G. R., Giblin, A. E., Nadelhoffer, K. J., & Laundre, J. A. (1995). Responses of arctic tundra to experimental and observed changes in climate. *Ecology*, *76*(3), 694–711. http://doi.org/10.2307/1939337

Chapin, F. S., Sturm, M., Serreze, M. C., McFadden, J. P., Key, J. R., Lloyd, A. H., McGuire, A. D., Rupp, T. S., Lynch, A. H., Schimel, J. P., Beringer, J., Chapman, W. L., Epstein, H. E., Euskirchen, E. S., Hinzman, L. D., Jia, G., Ping, C.-L., Tape, K. D., Thompson, C. D. C., Walker, D. A., & Welker, J. M. (2005). Role of land-surface changes in Arctic summer warming. *Science*, *310*(5748), 657–660. Retrieved from http://science.sciencemag.org/content/310/5748/657.abstract

Chapin, F. S., Zavaleta, E. S., Eviner, V. T., Naylor, R. L., Vitousek, P. M., Reynolds, H. L., Hooper, D. U., Lavorel, S., Sala, O. E., Hobbie, S. E., Mack, M. C., & Díaz, S. (2000). Consequences of changing biodiversity. *Nature*, *405*(6783), 234–242. https://doi.org/10.1038/35012241

Charity, S., Dudley, N., Oliveira, D., & Stolton, S. (Eds.). (2016). *Living Amazon Report 2016: A Regional Approach to Conservation in the Amazon*. Brasília, Brazil and Quito, Ecuador: WWF Living Amazon Initiative.

Chaudhary, A., Burivalova, Z., Koh, L. P., & Hellweg, S. (2016). Impact of forest management on species richness: global meta-analysis and economic trade-offs. *Scientific Reports*, *6*(April), 23954. http://doi.org/10.1038/srep23954

Cherry, J. E., Déry, S. J., Cheng, Y., Stieglitz, M., Jacobs, A. S., & Pan, F. (2014). Climate and hydrometeorology of the Toolik Lake Region and the Kuparuk River Basin: Past, present, and future. In J. E. Hobbie &

G. W. Kling (Eds.), *Alaska's Changing Arctic: Ecological Consequences for Tundra, Streams, and Lakes* (pp. 21–60). New York, USA: Oxford University Press.

Chesapeake Bay Program. (2017). Submerged Aquatic Vegetation. Retrieved from http://www.chesapeakebay.net/

Cheung, W. W., Watson, R., & Pauly, D. (2013). Signature of ocean warming in global fisheries catch. *Nature*, *497*(7449), 365–368. http://doi.org/10.1038/nature12156

Childs, J. E., & Seegar, W. S. (1986). Epidemiologic observations on infection with Toxoplasma gondii in three species of urban mammals from Baltimore, Maryland, USA. *International Journal of Zoonoses*, *13*(4), 249–261.

Choi, C. (2008). Tierra del Fuego: the beavers must die. *Nature*, *453*, 968. http://doi.org/10.1038/453968a

Cleland, E. E., Funk, J., & Allen., E. B. (2016). Coastal sage scrub. In E. Zavaleta & H. Mooney (Eds.), *Ecosystems of California* (pp. 429–448). Oakland, USA: University of California Press.

Clement, C., de Cristo-Araujo, M., D'Eeckenbrugge, G. C., Alves-Pereira, A., & Picanço-Rodrigues, D. (2016). 10,000 years of plant domestication: the origins of agrobiodiversity in indigenous Amazonia. In A. Casas, J. Torres-Guevara, & F. Parra (Eds.), *Domesticación en el Continente Americano Vol. 1. Investigación Manejo de Biodiversidad y Evolución Dirigida por las Culturas del Nuevo Mundo* (pp. 253–282). México D.F., México and Lima, Perú: Universidad Nacional Autónoma de México/Universidad Nacional Agraria La Molina.

Clement, C. R. (1999). 1492 and the loss of Amazonian crop genetic resources. II. Crop biogeography at contact. *Economic Botany*, *53*, 203–216. http://doi.org/10.1007/BF02866499

Clement, C. R. (2017). Panorama de los recursos genéticos de Brasil, con énfasis en la Amazonía. In A. Casas, J. Torres-Guevara, & F. Parra (Eds.), *Domesticación en el Continente Americano Vol. 2. Investigación para el Manejo Sustentable de Recursos Genéticos en el Nuevo Mundo* (pp. 27–42). México D.F. and Lima, Peru: Universidad Nacional Autónoma de México/Universidad Nacional Agraria La Molina/CONACYT.

Clement, C. R., de Cristo-Araújo, M., D'Eeckenbrugge, G. C., Pereira, A. A., & Picanço-Rodrigues, D. (2010). Origin and domestication of native Amazonian crops. *Diversity*, *2*(1), 72–106. http://doi.org/10.3390/d2010072

Cohen, A., & Carlton, J. (1998). Accelerating invasion rate in a highly invaded estuary. *Science*, *279*(5350), 555–558. http://doi.org/10.1126/science.279.5350.555

Collins, F. S. (1998). New goals for the U.S. Human Genome Project: 1998-2003. *Science*, *282*(5389), 682–689. http://doi.org/10.1126/science.282.5389.682

Comité Operativo para la Prevención el Control y la Erradicación de las Especies Exóticas Invasoras (COCEI). (2014). *Estrategia Nacional Integrada para la Prevención, el Control y/o Erradicación de las Especies Exóticas Invasoras*. Santiago, Chile: Ministerio del Medio Ambiente (MMA).

Committee on Taxonomy (COT). (2016). List of marine mammal species and subspecies. Retrieved January 14, 2017, from www.marinemammalscience.org

Conabio. (2014). Biodiversidad Mexicana. http://doi.org/10.2307/3503924

CONABIO. (2016). Sistema de Información sobre Especies Invasoras en México. Comisión Nacional para el Conocimiento y Uso de la Biodiversidad (Available on request).

CONABIO database. (n.d.). Retrieved from https//www.conabio.gob.mx

CONANP. (2017). Firma el Presidente Enrique Peña Nieto el decreto del Parque Nacional Revillagigedo. Retrieved from https://www.gob.mx/conanp/prensa/firma-el-presidente-enrique-pena-nieto-el-decreto-del-parque-nacional-revillagigedo-136220

Contreras, T. E., Figueroa, J. A., Abarca, L., & Castro, S. A. (2011). Fire regimen and spread of plants naturalized in central Chile. *Revista Chilena de Historia Natural*, *84*(3), 307–323. http://doi.org/10.4067/S0716-078X2011000300001

Cooney, P. B., & Kwak, T. J. (2010). Development of standard weight equations for Caribbean and Gulf of Mexico amphidromous fishes. *North American Journal of Fisheries Management*, *30*(5), 1203–1209. http://doi.org/10.1577/M10-058.1

Cooper, C. B., Loyd, K. A. T., Murante, T., Savoca, M., & Dickinson, J. (2012). Natural history traits associated with detecting mortality within residential bird communities: can citizen science provide insights? *Environmental Management*, *50*(1), 11–20. http://doi.org/10.1007/s00267-012-9866-x

Cornwell, W. K., Schwilk, D. W., & Ackerly, D. D. (2006). A trait-based test for habitat filtering: convex hull volume. *Ecology*, *87*(6), 1465–1471. http://doi.org/10.1890/0012-9658(2006)87%5B1465:ATTFHF%5D2.0.CO;2

Corporación Nacional Forestal (CONAF). (2017). *Análisis de la Afectación y Severidad de los Incendios Forestales ocurridos en Enero y Febrero de 2017 sobre los Usos de Suelo y los Ecosistemas Naturales presentes entre las Regiones de Coquimbo y Los Ríos de Chile*. Santiago, Chile. Retrieved from http://www.conaf.cl/tormenta_de_fuego-2017

Correa, F. (2011). Desencializando lo indígena. In G. R. Nemogá (Ed.), *Naciones Indígenas en los Estados Contemporáneos* (pp. 83–111). Bogotá, Colombia: Universidad Nacional de Colombia.

Correa, S. B., Araujo, J. K., Penha, J. M. F., Nunes da Cunha, C., Stevenson, P. R., & Anderson, J. P. (2015). Overfishing disrupts an ancient mutualism between frugivorous fishes and plants in Neotropical wetlands. *Biological Conservation*, *191*, 159–167. http://doi.org/10.1016/j.biocon.2015.06.019

Costanza, R., Pérez-Maqueo, O., Martinez, M. L., Sutton, P., Anderson, S. J., & Mulder, K. (2008). The value of coastal wetlands for hurricane protection. *Ambio*, *37*(4), 241–248. http://doi.org/10.1579/0044-7447(2008)37%5B241:TVOCWF%5D2.0.CO;2

Costello, C., Ovando, D., Clavelle, T., Strauss, C. K., Hilborn, R., Melnychuk, M. C., Branch, T. A., Gaines, S. D., Szuwalski, C. S., Cabral, R. B., Rader, D. N., & Leland, A. (2016). Global fishery prospects under contrasting management regimes. *PNAS*, *113*(18), 5125–5129. https://doi.org/10.1073/pnas.1520420113

Costello, E. K., Halloy, S. R. P., Reed, S. C., Sowell, P., & Schmidt, S. K. (2009). Fumarole-supported islands of biodiversity within a hyperarid, high-elevation landscape on Socompa volcano, Puna de Atacama, Andes. *Applied and Environmental Microbiology*, *75*(3), 735–747. http://doi.org/10.1128/AEM.01469-08

Crandall, K. A., & Buhay, J. E. (2008). Global diversity of crayfish (Astacidae, Cambaridae, and Parastacidae - Decapoda) in freshwater. *Hydrobiologia*, *595*(1), 295–301. http://doi.org/10.1007/s10750-007-9120-3

Crawford, A. J., Alonso, R., Jaramillo, C. A. A., Sucre, S., & Ibáñez, R. D. (2011). DNA barcoding identifies a third invasive species of *Eleutherodactylus* (Anura: Eleutherodactylidae) in Panama City, Panama. *Zootaxa*, *67*(2890), 65–67.

Crimmins, S. M., Dobrowski, S. Z., Greenberg, J. A., Abatzoglou, J. T., & Mynsberge, A. R. (2011). Changes in climatic water balance drive downhill shifts in plant species' optimum elevations. *Science*, *331*(6015), 324–327. http://doi.org/10.1126/science.1199040

Crisp, Laffan, Linder, & Monro. (2001). Endemism in the Australian flora. *Journal of Biogeography*, *28*(2), 183–198. http://doi.org/10.1046/j.1365-2699.2001.00524.x

Crits-Christoph, A., Gelsinger, D. R., Ma, B., Wierzchos, J., Ravel, J., Davila, A., Casero, M. C., & DiRuggiero, J. (2016). Functional interactions of archaea, bacteria and viruses in a hypersaline endolithic community. *Environmental Microbiology*, *18*(6), 2064–2077. https://doi.org/10.1111/1462-2920.13259

Cuesta, F., Muriel, P., Llambí, L. D., Halloy, S., Aguirre, N., Beck, S., Carilla, J., Meneses, R. I., Cuello, S., Grau, A., Gámez, L. E., Irazábal, J., Jácome, J., Jaramillo, R., Ramírez, L., Samaniego, N., Suárez-Duque, D., Thompson, N., Tupayachi, A., Viñas, P., Yager, K., Becerra, M. T., Pauli, H., & Gosling, W. D. (2017). Latitudinal and altitudinal patterns of plant community diversity on mountain summits across the tropical Andes. *Ecography*, *40*(12), 1381–1394. https://doi.org/10.1111/ecog.02567

Cushman, S. A. (2006). Effects of habitat loss and fragmentation on amphibians: A review and prospectus. *Biological Conservation*, *128*(2), 231–240. http://doi.org/10.1016/j.biocon.2005.09.031

da Silva, F. R., Candeira, C. P., & Rossa-Feres, D. C. (2012). Dependence of anuran diversity on environmental descriptors in farmland ponds. *Biodiversity and Conservation*, *21*(6), 1411–1424. http://doi.org/10.1007/s10531-012-0252-z

da Silva, F. R., Gibbs, J. P., & Rossa-Feres, D. C. (2011). Breeding habitat and landscape correlates of frog diversity and abundance in a tropical agricultural landscape. *Wetlands*, *31*(6), 1079–1087. http://doi.org/10.1007/s13157-011-0217-0

Dahl, T. E. (1990). *Wetland Losses in the United States 1780's to 1980's*. Washington, D.C., USA: Department of the Interior, Fish and Wildlife Service.

Dahl, T. E. (2011). *Status and Trends of Wetlands in the Conterminous United States 2004 to 2009*. Washington D.C., USA: U.S. Department of the Interior; Fish and Wildlife Service.

Dai, D. (2011). Racial/ethnic and socioeconomic disparities in urban green space accessibility: Where to intervene? *Landscape and Urban Planning*, *102*(4), 234–244. http://doi.org/10.1016/j.landurbplan.2011.05.002

Dangles, O., Carpio, C., Barragan, A. R., Zeddam, J.-L., & Silvain, J.-F. (2008). Temperature as a key driver of ecological sorting among invasive pest species in the tropical Andes. *Ecological Applications*, *18*(7), 1795–1809. http://doi.org/10.1890/07-1638.1

Daniels, A. E., & Cumming, G. S. (2008). Conversion or conservation? Understanding wetland change in northwest Costa Rica. *Ecological Applications*, *18*(1), 49–63. http://doi.org/10.1890/06-1658.1

Daniels, L. D., & Veblen, T. T. (2004). Spatiotemporal influences of climate on altitudinal treeline in northern Patagonia. *Ecology*, *85*(5), 1284–1296. http://doi.org/10.1890/03-0092

Danielsen, F., Beukema, H., Burgess, N. D., Parish, F., Buhl, C. A., Donald, P. F., Murdiyarso, D., Phalan, B., Reijnders, L., Struebig, M., & Fitzherbert, E. B. (2009). Biofuel plantations on forested lands: double jeopardy for biodiversity and climate. *Conservation Biology*, *23*(2), 348–358. https://doi.org/10.1111/j.1523-1739.2008.01096.x

Darrigran, G., Damborenea, C., Drago, E. C., Ezcurra de Drago, I., Paira, A., & Archuby, F. (2012). Invasion process of *Limnoperna fortunei* (Bivalvia: Mytilidae): the case of Uruguay River and emissaries of the Esteros del Iberá Wetland, Argentina. *Zoología*, *29*(6), 531–539. http://doi.org/10.1590/S1984-46702012000600004

Darrigran, G., & Ezcurra de Drago, I. (2000). Invasion of *Limnoperna fortunei* (Dunker, 1857) (Bivalvia: Mytilidae) in America. *The Nautilus*, *114*, 69–73. http://doi.org/10.1007/s10530-005-0331-0

Davidson, E. A., de Araújo, A. C., Artaxo, P., Balch, J. K., Brown, I. F., Bustamante, M. M., Coe, M. T., DeFries, R. S., Keller, M., Longo, M., Munger, J. W., Schroeder, W., Soares-Filho, B. S., Souza, C. M., & Wofsy, S. C. (2012). The Amazon basin in transition. *Nature*, *481*(7381), 321–328. https://doi.org/10.1038/nature10717

Davidson, L. N. K., Krawchuk, M. A., & Dulvy, N. K. (2016). Why have global shark and ray landings declined: Improved management or overfishing? *Fish and Fisheries*, *17*(2), 438–458. http://doi.org/10.1111/faf.12119

Davies, T. J., & Buckley, L. B. (2011). Phylogenetic diversity as a window into the evolutionary and biogeographic histories of present-day richness gradients for mammals. *Philosophical Transactions of the Royal Society B: Biological Sciences*, *366*(1576), 2414–2425. http://doi.org/10.1098/rstb.2011.0058

Davis, F. W., Baldocchi, D. D., & Tyler, C. M. (2016). Oak woodlands. In H. Mooney & E. Zavaleta (Eds.), *Ecosystems of California* (pp. 509–529). Oakland, USA: University of California Press.

De Jong, W. (1996). Swidden-fallow agroforestry in Amazonia: diversity at close distance. *Agroforestry Systems*, *34*, 277–290. http://doi.org/10.1007/BF00046928

De la Cruz, R. (2011). Conocimientos tradicionales, biodiversidad y derechos de propiedad intelectual, patentes. In G. R. Nemogá (Ed.), *Naciones Indígenas en los Estados Contemporáneos* (pp. 211–230). Bogotá, Colombia: Universidad Nacional de Colombia.

De la Cruz, R., Muyuy Jacanamejoy, G., Viteri Gualinga, A., Flores, G., Humpire, J. G., Mirabal DÌaz, J. G., & Guimaraez, R. (2005). *Elementos para la Protección sui generis de los Conocimientos Tradicionales Colectivos e Integrales desde la Perspectiva Indigena*. Caracas, Venezuela: The Corporacion Andina de Fomento (CAF)/Comunidad Andina (CAN).

de Oliveira, G., Araújo, M. B., Rangel, T. F., Alagador, D., & Diniz-Filho, J. A. F. (2012). Conserving the Brazilian semiarid (Caatinga) biome under climate change. *Biodiversity and Conservation*, *21*(11), 2913–2926. http://doi.org/10.1007/s10531-012-0346-7

de Vasconcellos Pegas, F., & Castley, J. G. (2016). Private reserves in Brazil: Distribution patterns, logistical challenges, and conservation contributions. *Journal for Nature Conservation*, *29*, 14–24. http://doi.org/10.1016/j.jnc.2015.09.007

DeClerck, F. A. J., Chazdon, R., Holl, K. D., Milder, J. C., Finegan, B., Martinez-Salinas, A., Imbach, P., Canet, L., & Ramos, Z. (2010). Biodiversity conservation in human-modified landscapes of Mesoamerica: Past, present and future. *Biological Conservation*, *143*, 2301–2313.

https://doi.org/10.1016/j.biocon.2010.03.026

Dee, L. E., Horii, S. S., & Thornhill, D. J. (2014). Conservation and management of ornamental coral reef wildlife: Successes, shortcomings, and future directions. *Biological Conservation*, *169*, 225–237.

Dee, L. E., Miller, S. J., Peavey, L. E., Bradley, D., Gentry, R. R., Startz, R., Gaines, S. D., & Lester, S. E. (2016). Functional diversity of catch mitigates negative effects of temperature variability on fisheries yields. *Proceedings of the Royal Society of London B: Biological Sciences*, *283*(1836), 20161435. https://doi.org/http://dx.doi.org/10.1098/rspb.2016.1435

Del Hoyo, J., Elliott, A., & Sargatal, J. (Eds.). (1992a). *Handbook of the Birds of the World, Volume 1: Ostrich to Ducks*. Barcelona, Spain: Lynx Edicions.

Del Hoyo, J., Elliott, A., & Sargatal, J. (Eds.). (1992b). *Handbook of the Birds of the World, Volume 3: Hoatzin to Auks*. Barcelona, Spain: Lynx Edicions.

Delis, P. R., Mushinsky, H. R., & McCoy, E. D. (1996). Decline of some west-central Florida anuran populations in response to habitat degradation. *Biodiversity and Conservation*, *5*(12), 1579–1595. http://doi.org/10.1007/BF00052117

Denslow, J. S., & DeWalt, S. J. (2008). Exotic plant invasions in tropical forests: patterns and hypotheses. In W. Carson & S. Schnitze (Eds.), *Tropical forest community ecology* (pp. 409–426). Oxford, UK: Wiley-Blackwell.

Derlindati, E. J., Romano, M. C., Cruz, N. N., Barisón, C., Arengo, F., & Barberis, I. M. (2014). Seasonal activity patterns and abundance of Andean flamingo (*Phoenicoparrus andinus*) at two contrasting wetlands in Argentina. *Ornitología Neotropical*, *25*(3), 317–331.

Díaz-Betancourt, M., Ghermandi, L., Ladio, A., López-Moreno, I. R., Raffaele, E., & Rapoport, E. H. (1999). Weeds as a source for human consumption. A comparison between tropical and temperate Latin America. *Revista de Biologia Tropical*, *47*(3), 329–338.

Díaz, I. A., & Armesto, J. J. (2003). La conservación de las aves silvestres en ambientes urbanos de Santiago. *Revista Ambiente Y Desarrollo de CIPMA*, *19*(2), 31–38.

Diaz, R. J., & Rosenberg, R. (2008). Spreading dead zones and consequences for marine ecosystems. *Science*, *321*(5891), 926–929. http://doi.org/10.1126/science.1156401

Diffenbaugh, N. S., Swain, D. L., & Touma, D. (2015). Anthropogenic warming has increased drought risk in California. *PNAS*, *112*(13), 3931–3936. http://doi.org/10.1073/pnas.1422385112

Dillon, M. O., Leiva González, S., Zapata Cruz, M., Lezama Asencio, P., & Quipuscoa Silvestre, V. (2011). Floristic checklist of the Peruvian lomas formations. *Arnaldoa*, *18*(1), 7–32.

Dinerstein, E., Olson, D. M., Graham, D. J., Webster, A. L., Primm, S. A., Bookbinder, M. P., & Ledec, G. (1995). *A Conservation Assessment of the Terrestrial Ecoregions of Latin America and the Caribbean*. Washington, D.C., USA: The World Bank and World Wildlife Fund. Retrieved from http://documents.worldbank.org/curated/en/1995/09/697067/conservation-assessment-terrestrial-ecoregions-latin-america-caribbean

Dirzo, R., & Raven, P. (2003). Global state of biodiversity and loss. *Annual Reivew of the Environment and Resources*, *28*, 137–167.

Díaz, S., & Cabido, M. (2001). Vive la différence: plant functional diversity matters to ecosystem processes. *Trends in Ecology & Evolution*, *16*(11), 646–655. http://doi.org/DOI: http://dx.doi.org/10.1016/S0169-5347(01)02283-2

Dolanc, C. R., Safford, H. D., Dobrowski, S. Z., & Thorne, J. H. (2014). Twentieth century shifts in abundance and composition of vegetation types of the Sierra Nevada, CA, US. *Applied Vegetation Science*, *17*(3), 442–455. http://doi.org/10.1111/avsc.12079

Dolanc, C. R., Safford, H. D., Thorne, J. H., & Dobrowski, S. Z. (2014). Changing forest structure across the landscape of the Sierra Nevada, CA, USA, since the 1930s. *Ecosphere*, *5*(8), art101. http://doi.org/10.1890/ES14-00103.1

Domínguez, E., Bahamonde, N., & Muñoz-Escobar, C. (2012). Efectos de la extracción de turba sobre la composición y estructura de una turbera de Sphagnum explotada y abandonada hace 20 años, Chile. *Anales Del Instituto de La Patagonia*, *40*(2), 37–45. http://doi.org/10.4067/S0718-686X2012000200003

Donohue, R. J., Roderick, M. L., McVicar, T. R., & Farquhar, G. D. (2013). Impact of CO2 fertilization on maximum foliage cover across the globe's warm, arid environments. *Geophysical Research*

Letters, *40*(12), 3031–3035. http://doi.org/10.1002/grl.50563

Dornelas, M., Gotelli, N. J., McGill, B., Shimadzu, H., Moyes, F., Sievers, C., & Magurran, A. E. (2014). Assemblage time series reveal biodiversity change but not systematic loss. *Science*, *344*(6181), 296–299. http://doi.org/10.1126/science.1248484

Doroff, A., & Burdin, A. (2015). *Enhydra lutris*. The IUCN Red List of Threatened Species 2015: e.T7750A21939518. Retrieved from http://dx.doi.org/10.2305/IUCN.UK.2015-2.RLTS.T7750A21939518.en.

Duarte, C. M. (1995). Submerged aquatic vegetation in relation to different nutrient regimes. *Ophelia*, *41*(1), 87–112. http://doi.org/10.1080/00785236.1995.10422039

Duboc, E., Costa, C. J., Veloso, R. F., Oliveira, L. S., & Paludo, A. (2007). *Panorama Atual da Produção de Carvão Vegetal no Brasil e no Cerrado. Documentos - Embrapa Cerrados No. 197*. Retrieved from http://www.cpac.embrapa.br

Duffy, J. E., Lefcheck, J. S., Stuart-Smith, R. D., Navarrete, S. A., & Edgar, G. J. (2016). Biodiversity enhances reef fish biomass and resistance to climate change. *PNAS*, *113*(22), 6230–6235. http://doi.org/10.1073/pnas.1524465113

Dulvy, N. K., Fowler, S. L., Musick, J. A., Cavanagh, R. D., Kyne, P. M. ., Harrison, L. R., Carlson, J. K., Davidson, L. N., Fordham, S. V, Francis, M. P., Pollock, C. M., Simpfendorfer, C. A., Burgess, G. H., Carpenter, K. E., Compagno, L. J., Ebert, D. A., Gibson, C., Heupel, M. R., Livingstone, S. R., Sanciangco, J. C., Stevens, J. D., Valenti, S., & White, W. T. (2014). Extinction risk and conservation of the world's sharks and rays. *ELife*, *3*, e00590. https://doi.org/10.7554/eLife.00590

Dunic, J. C. (2016). *Contextualizing Local Biodiversity Change in the Face of Human Impacts.* Masters Dissertation, 415, University of Massachusetts Boston. Retrieved from https://scholarworks.umb.edu/masters_theses/415

Dunn, R. R., Agosti, D., Andersen, A. N., Arnan, X., Bruhl, C. A., Cerdá, X., Ellison, A. M., Fisher, B. L., Fitzpatrick, M. C., Gibb, H., Gotelli, N. J., Gove, A. D., Guenard, B., Janda, M., Kaspari, M., Laurent, E. J., Lessard, J. P., Longino, J. T., Majer, J. D., Menke, S. B., McGlynn, T. P., Parr, C. L., Philpott, S. M., Pfeiffer, M., Retana, J., Suarez, A. V., Vasconcelos, H. L., Weiser, M. D., & Sanders, N. J. (2009). Climatic drivers of hemispheric asymmetry in global patterns of ant species richness. *Ecology Letters*, *12*(4), 324–333. https://doi.org/10.1111/j.1461-0248.2009.01291.x

Durán, A., Gryzenhout, M., Slippers, B., Ahumada, R., Rotella, A., Flores, F., Wingfield, B. D., & Wingfield, M. J. (2008). *Phytophthora pinifolia* sp. nov. associated with a serious needle disease of *Pinus radiata* in Chile. *Plant Pathology*, *57*(4), 715–727. https://doi.org/10.1111/j.1365-3059.2008.01893.x

Durigan, G., & Ratter, J. A. (2016). The need for a consistent fire policy for Cerrado conservation. *Journal of Applied Ecology*, *53*(1), 11–15. http://doi.org/10.1111/1365-2664.12559

Dyer, E. E., Cassey, P., Redding, D. W., Collen, B., Franks, V., Gaston, K. J., Jones, K. E., Kark, S., Orme, C. D. L., & Blackburn, T. M. (2017). The global distribution and drivers of alien bird species richness. *PLoS Biol*, *15*(1), e2000942. https://doi.org/10.1371/journal.pbio.2000942

Early, R., Bradley, B. A., Dukes, J. S., Lawler, J. J., Olden, J. D., Blumenthal, D. M., Gonzalez, P., Grosholz, E. D., Ibañez, I., Miller, L. P., Sorte, C. J. B., & Tatem, A. J. (2016). Global threats from invasive alien species in the twenty-first century and national response capacities. *Nature Communications*, *7*, art12485. https://doi.org/10.1038/ncomms12485

Eberhardt, L. L., & O'Shea, T. J. (1995). Integration of manatee life history data and population modeling. In T. J. O'Shea, B. B. Ackerman, & H. F. Percival (Eds.), *Population Biology of the Florida Manatee* (pp. 269–273). Washington D.C., USA: National Biological Service, Information and Technology Report.

Eclesia, R. P., Jobbagy, E. G., Jackson, R. B., Biganzoli, F., & Piñeiro, G. (2012). Shifts in soil organic carbon for plantation and pasture establishment in native forests and grasslands of South America. *Global Change Biology*, *18*(10), 3237–3251. http://doi.org/10.1111/j.1365-2486.2012.02761.x

Edwards, T., Schwalbe, C. R., Swann, D. E., & Goldberg, C. S. (2004). Implications of anthropogenic landscape change on inter-population movements of the desert tortoise (*Gopherus agassizii*). *Conservation Genetics*, *5*(4), 485–499. http://doi.org/10.1023/B:COGE.0000041031.58192.7c

Eisemberg, C. C., Machado Balestra, R. A., Famelli, S., Pereira, F. F., Diniz Bernardes, V. C., & Vogt, R. C. (2016). Vulnerability of giant South American turtle (*Podocnemis expansa*) nesting habitat to climate-change-induced alterations to fluvial cycles. *Tropical Conservation Science*, (October-

December issue), 1–12. http://doi.org/10.1177/1940082916667139

Elahi, R., O'Connor, M. I., Byrnes, J. E. K., Dunic, J., Eriksson, B. K., Hensel, M. J. S., & Kearns, P. J. (2015). Recent trends in local-scale marine biodiversity reflect community structure and human impacts. *Current Biology*, *25*(14), 1938–1943. http://doi.org/10.1016/j.cub.2015.05.030

Elliott, G. P. (2011). Influences of 20th-century warming at the upper tree line contingent on local-scale interactions: evidence from a latitudinal gradient in the Rocky Mountains, USA. *Global Ecology and Biogeography*, *20*(1), 46–57.

Ellison, A. M., & Farnsworth, E. J. (1996). Anthropogenic disturbance of Caribbean mangrove ecosystems: Past impacts, present trends, and future predictions. *Biotropica*, *28*(4), 549–565. http://doi.org/10.2307/2389096

Elmendorf, S. C., Henry, G. H. R., Hollister, R. D., Björk, R. G., Boulanger-Lapointe, N., Cooper, E. J., Cornelissen, J. H. C., Day, T. A., Dorrepaal, E., Elumeeva, T. G., Gill, M., Gould, W. A., Harte, J., Hik, D. S., Hofgaard, A., Johnson, D. R., Johnstone, J. F., Jónsdóttir, I. S., Jorgenson, J. C., Klanderud, K., Klein, J. A., Koh, S., Kudo, G., Lara, M., Lévesque, E., Magnússon, B., May, J. L., Mercado-Díaz, J. A., Michelsen, A., Molau, U., Myers-Smith, I. H., Oberbauer, S. F., Onipchenko, V. G., Rixen, C., Schmidt, N. M., Shaver, G. R., Spasojevic, M. J., Þórhallsdóttir, Þ. E., Tolvanen, A., Troxler, T., Tweedie, C. E., Villareal, S., Wahren, C.-H., Walker, X., Webber, P. J., Welker, J. M., & Wipf, S. (2012). Plot-scale evidence of tundra vegetation change and links to recent summer warming. *Nature Climate Change*, *2*, 453–457. https://doi.org/10.1038/nclimate1465

Elsner, W. K., & Jorgenson, J. C. (2009). White spruce seedling (*Picea glauca*) discovered north of the Brooks Range along Alaska's Dalton Highway. *Arctic*, *62*(3), 342–344.

Elven, R., Murray, D. F., Razzhivin, V. Y., & Yurtsev, B. A. (2011). Annotated checklist of the panarctic flora (PAF): vascular plants. Retrieved from http://geo.abds.is/geonetwork/srv/eng/catalog.search -/metadata/ad3b16c9-9a29-460e-a515-d0a48e6d88ff

Emperaire, L., & Peroni, N. (2007). Traditional management of agrobiodiversity in Brazil: A case study of manioc. *Human Ecology*, *35*, 761–768.

Enquist, B. J., Condit, R., Peet, R. K., Schildhauer, M., & Thiers, B. (2016). The Botanical Information and Ecology Network (BIEN): Cyberinfrastructure for an integrated botanical information network to investigate the ecological impacts of global climate change on plant biodiversity. http://doi.org/10.7287/peerj.preprints.2615v2.

Epstein, H. E., Raynolds, M. K., Walker, D. A., Bhatt, U. S., Tucker, C. J., & Pinzon, J. E. (2012). Dynamics of aboveground phytomass of the circumpolar Arctic tundra during the past three decades. *Environmental Research Letters*, *7*(1), 15506.

Ernst, O., & Siri-Prieto, G. (2009). Impact of perennial pasture and tillage systems on carbon input and soil quality indicators. *Soil and Tillage Research*, *105*(2), 260–268. http://doi.org/10.1016/j.still.2009.08.001

Eskew, E. A., Price, S. J., & Dorcas, M. E. (2012). Effects of river-flow regulation on anuran occupancy and abundance in riparian zones. *Conservation Biology*, *26*(3), 504–512. http://doi.org/10.1111/j.1523-1739.2012.01842.x

Espinosa, F. J., & Vibrans, H. (2009). The need for a national weed management strategy. In T. van Devender, F. J. Espinosa-García, B. L. Harper-Lore, & T. Hubbard (Eds.), *Invasive Plants on the Move: Controlling them in North America* (pp. 23–32). Tucson, USA: Arizona-Sonora Desert Museum.

Estay, S. A. (2016). Invasive insects in the mediterranean forests of Chile. In T. D. Paine & F. Lieutier (Eds.), *Insects and Diseases of Mediterranean Forest Systems* (pp. 379–396). New York, USA: Springer. http://doi.org/10.1007/978-3-319-24744-1_13

Estevez, R. A., Squeo, F. A., Arancio, G., & Erazo, M. B. (2010). Production of charcoal from native shrubs in the Atacama Region, Chile. *Gayana Botánica*, *67*(2), 213–222.

Eva, H. D., Belward, A. S., De Miranda, E. E., Di Bella, C. M., Gond, V., Huber, O., Jones, S., Sgrenzaroli, M., & Fritz, S. (2004). A land cover map of South America. *Global Change Biology*, *10*(5), 731–744. https://doi.org/10.1111/j.1529-8817.2003.00774.x

Evans, P., & Brown, C. D. (2017). The boreal-temperate forest ecotone response to climate change. *Environmental Reviews*, *25*(4), 423–431. http://doi.org/10.1139/er-2017-0009

Ewing, H. A., Tuininga, A. R., Groffman, P. M., Weathers, K. C., Fahey, T. J., Fisk, M. C., Bohlen, P. J., & Suarez,

E. (2015). Earthworms reduce biotic 15-nitrogen retention in northern hardwood forests. *Ecosystems*, *18*(2), 328–342. https://doi.org/10.1007/s10021-014-9831-z

Faeth, S. H., Marussich, W. A., Shochat, E., & Warren, P. S. (2005). Trophic dynamics in urban communities. *BioScience*, *55*(5), 399–407. http://doi.org/10.1641/0006-3568(2005)055%5B0399:TDIUC%5D2.0.CO;2

Faggi, A., & Perepelizin, P. V. (2006). Riqueza de aves a lo largo de un gradiente de urbanización en la ciudad de Buenos Aires. *Revista Del Museo Argentino de Ciencias Naturales*, *8*(2), 289–297.

Fahrig, L. (2003). Effects of habitat fragmentation on biodiversity. *Annual Review of Ecology, Evolution, and Systematics*, *34*(1), 487–515. http://doi.org/10.1146/annurev.ecolsys.34.011802.132419

Fahrig, L., Pedlar, J. H., Pope, S. E., Taylor, P. D., & Wegner, J. F. (1995). Effect of road traffic on amphibian density. *Biological Conservation*, *73*(3), 177–182. http://doi.org/10.1016/0006-3207(94)00102-V

Faith, D. P. (1992). Conservation evaluation and phylogenetic diversity. *Biological Conservation*, *61*(1), 1–10. http://doi.org/10.1016/0006-3207(92)91201-3

Fajardo, A., & McIntire, E. J. B. (2012). Reversal of multicentury tree growth improvements and loss of synchrony at mountain tree lines point to changes in key drivers. *Journal of Ecology*, *100*(3), 782–794. http://doi.org/10.1111/j.1365-2745.2012.01955.x

Falvey, M., & Garreaud, R. D. (2009). Regional cooling in a warming world: recent temperature trends in the southeast Pacific and along the west coast of subtropical South America (1979-2006). *Journal of Geophysical Research: Atmospheres*, *114*(D4). http://doi.org/10.1029/2008JD010519

FAO. (2007). *The State of Food and Agriculture 2007*. Rome, Italy. Retrieved from http://www.fao.org/docrep/010/a1200e/a1200e00.htm

FAO. (2011). *The State of Forest Resources- a Regional Analysis. In State of the World's Forests*. *Food and Agriculture Organization of the United Nations.* Rome, Italy. Retrieved from http://www.fao.org/docrep/013/i2000e/i2000e00.htm

FAO. (2012). The State of World Fisheries and Aquaculture 2012. *Aquaculture Newsletter*, 209. Retrieved from http://www.fao.org/3/a-i2727e.pdf

FAO. (2016). *The State of World Fisheries and Aquaculture 2016: Contributing to Food Security and Nutrition for All*. *Food and Agriculture Organization of the United Nations*. Rome, Italy. Retrieved from http://www.fao.org/3/a-i5555e.pdf

Fargione, J., Tilman, D., Dybzinski, R., Lambers, J. H. R., Clark, C., Harpole, W. S., Knops, J. M. H., Reich, P. B., & Loreau, M. (2007). From selection to complementarity: shifts in the causes of biodiversity-productivity relationships in a long-term biodiversity experiment. *Proceedings of the Royal Society Biological Sciences*, *274*(1611), 871–876. https://doi.org/10.1098/rspb.2006.0351

Farley, K. A., Bremer, L. L., Harden, C. P., & Hartsig, J. (2013). Changes in carbon storage under alternative land uses in biodiverse Andean grasslands: implications for payment for ecosystem services. *Conservation Letters*, *6*(1), 21–27. http://doi.org/10.1111/j.1755-263X.2012.00267.x

Farley, K. A., Jobbágy, E. G., & Jackson, R. B. (2005). Effects of afforestation on water yield: a global synthesis with implications for policy. *Global Change Biology*, *11*(10), 1565–1576. http://doi.org/10.1111/j.1365-2486.2005.01011.x

Farley, K. A., Kelly, E. F., & Hofstede, R. G. M. (2004). Soil organic carbon and water retention after conversion of grasslands to pine plantations in the Ecuadorian Andes. *Ecosystems*, *7*(7), 729–739. http://doi.org/10.1007/s10021-004-0047-5

Federal Provincial and Territorial Governments of Canada. (2010). *Canadian Biodiversity: Ecosystem Status and Trends 2010*. Ottawa, Canada: Canadian Councils of Resource Ministers. Retrieved from http://www.biodivcanada.ca/A519F000-8427-4F8C-9521-8A95AE287753/EN_CanadianBiodiversity_FULL.pdf

Federal Register. (2016). *Indian Entities Recognized and Eligible to Receive Services From the United States Bureau of Indian Affairs*. Bureau of Indian Affairs. Retrieved from https://www.gpo.gov/fdsys/pkg/FR-2016-01-29/pdf/2016-01769.pdf

Feeley, K. J., Silman, M. R., Bush, M. B., Farfan, W., Cabrera, K. G., Malhi, Y., Meir, P., Revilla, N. S., Quisiyupanqui, M. N. R., & Saatchi, S. (2011). Upslope migration of Andean trees. *Journal of Biogeography*, *38*(4), 783–791. https://doi.org/10.1111/j.1365-2699.2010.02444.x

Fehlenberg, V., Baumann, M., Gasparri, N. I., Piquer-Rodriguez, M., Gavier-Pizarro, G., & Kuemmerle, T.

(2017). The role of soybean production as an underlying driver of deforestation in the South American Chaco. *Global Environmental Change*, *45*(April), 24–34. http://doi.org/10.1016/j.gloenvcha.2017.05.001

Fernandez, A. B., Rasuk, M. C., Visscher, P. T., Contreras, M., Novoa, F., Poire, D. G., Patterson, M. M., Ventosa, A., & Farias, M. E. (2016). Microbial diversity in sediment ecosystems (evaporites domes, microbial mats, and crusts) of hypersaline Laguna Tebenquiche, Salar de Atacama, Chile. *Frontiers in Microbiology*, *7*, 1284. https://doi.org/10.3389/fmicb.2016.01284

Ferrusquía-Villafranca, I., González-Guzmán, L. I., & Cartron, J. E. (2005). Northern Mexico's landscape, part I: the physical setting and constraints on modeling biotic evolution. In J. E. Cartron, G. Ceballos, & R. S. Felger (Eds.), *Biodiversity, Ecosystems, and Conservation in Northern Mexico* (pp. 11–38). New York, USA: Oxford University Press.

Fields, J. P., Belnap, J., Breshears, D. D., Neff, J. ., Okin, G. S., Whicker, J. J., Painter, T. H., Ravi, S., Reheis, M. C., & Reynolds, R. L. (2009). The ecology of dust. *Frontiers in Ecology and the Environment*, *8*(8), 423–430. https://doi.org/10.1890/090050

Figueroa, J. A., Teillier, S., & Castro, S. A. (2011). Diversity patterns and composition of native and exotic floras in central Chile. *Acta Oecologica*, *37*(2), 103–109. http://doi.org/10.1016/j.actao.2011.01.002

Fish, M. R., Lombana, A., & Drews., C. (2009). *Regional Climate Projections: Climate Change and Marine Turtles in the Wider Caribbean*. San Jose, Costa Rica: WWF report. Retrieved from http://awsassets.panda.org/downloads/climate_change_and_marine_turtles_in_the_wider_caribbean_1.pdf

Fisheries and Oceans Canada. (2017). Action plan for the Northern and Southern resident Killer Whale (*Orcinus orca*) in Canada. Species at Risk Action Plan Series. Ottawa, Canada: Fisheries and Oceans Canada. Retrieved fromhttp://www.registrelep-sararegistry.gc.ca/virtual_sara/files/plans/Ap-ResidentKillerWhale-v00-2017Mar-Eng.pdf

Fitzpatrick, S. M., & Keegan, W. F. (2007). Human impacts and adaptations in the Caribbean Islands: an historical ecology approach. *Earth and Environmental Science Transactions of the Royal Society of Edinburgh*, *98*(1), 29–45.

Fjeldså, J. (2002). *Polylepis* forests - vestiges of a vanishing ecosystem in the Andes. *Ecotropica*, *8*(2), 111–123.

Fjeldså, J., & Rahbek, C. (2006). Diversification of tanagers, a species rich bird group, from lowlands to montane regions of South America. *Integrative and Comparative Biology*, *46*(1), 72–81. http://doi.org/10.1093/icb/icj009

Flather, C. H., Brady, S. J., & Knowles, M. S. (1999). Wildlife Resource Trends in the United States: A Technical Document Supporting the 2000 RPA Assessment. Gen. Tech. Rep. RMRS-GTR-33. Fort Collins, USA: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

Flecker, A. S., McIntyre, P. B., Moore, J. W., Anderson, J. T., Taylor, B. W., & Hall Jr., R. O. (2010). Migratory fishes as material and process subsidies in riverine ecosystems. *American Fisheries Society Symposium*, *73*(2), 559–592. Retrieved from http://www.dartmouth.edu/~btaylor/index/Publications_files/flecker_etal_2010_AFSS_migrator-fishes.pdf

Flores-Villela, O., & Gerez, P. (1994). *Biodiversidad y Conservación en México: Vertebrados, Vegetación y Uso del Suelo* (2nd ed.). México D.F., México: Universidad Nacional Autónoma de México.

Fonseca, C. R., Ganade, G., Baldissera, R., Becker, C. G., Boelter, C. R., Brescovit, A. D., Campos, L. M., Fleck, T., Fonseca, V. S., Hartz, S. M., Joner, F., Käffer, M. I., Leal-Zanchet, A. M., Marcelli, M. P., Mesquita, A. S., Mondin, C. A., Paz, C. P., Petry, M. V., Piovensan, F. N., Putzke, J., Stranz, A., Vergara, M., & Vieira, E. M. (2009). Towards an ecologically-sustainable forestry in the Atlantic Forest. *Biological Conservation*, *142*(6), 1209–1219. https://doi.org/10.1016/j.biocon.2009.02.017

Forbes, B. C., & Jefferies, R. L. (1999). Revegetation of disturbed arctic sites: constraints and applications. *Biological Conservation*, *88*(1), 15–24. http://doi.org/10.1016/S0006-3207(98)00095-0

Ford, A., & Nigh, R. (2015). *Maya Forest Garden: Eight Millennia of Sustainable Cultivation of the Tropical Woodlands*. New York, USA: Routledge Taylor & Francis Group.

Ford, M. A., Cahoon, D., & Lynch, J. C. (1999). Restoring marsh elevation in a rapidly subsiding salt marsh by thin layer deposition of dredged material. *Ecological Engineering*, *12*(3), 189–205.

Forero Ulloa, F. E., Cely R., Germán, E., & Palacios Pacheco, L. S. (2015). *Dinámica del Páramo como Espacio para la Captura de Carbono*. Tunja, Colombia: Universidad Pedagógica y Tecnológica de Colombia (UPTC).

Forest, F., Grenyer, R., Rouget, M., Davies, T. J., Cowling, R. M., Faith, D. P., Balmford, A., Manning, J. C., Proches, S., Bank, M. van der, Reeves, G., Hedderson, T. A. J., & Savolainen, V. (2007). Preserving the evolutionary potential of floras in biodiversity hotspots. *Nature*, *445*(7129), 757–760. https://doi.org/10.1038/nature05587

Forister, M. L., & Shapiro, A. M. (2003). Climatic trends and advancing spring flight of butterflies in lowland California. *Global Change Biology*, *9*(7), 1130–1135. http://doi.org/10.1046/j.1365-2486.2003.00643.x

Formica, A., Farrer, E. C., Ashton, I. W., & Suding, K. N. (2014). Shrub expansion over the past 62 years in Rocky Mountain alpine tundra: possible causes and consequences. *Arctic, Antarctic, and Alpine Research*, *46*(3), 616–631.

Foster, R. B., & Hubbell, S. P. (1990). The floristic composition of the Barro Colorado Island forest. In A. H. Gentry (Ed.), *Four Neotropical Rainforests* (pp. 85–98). New Haven, USA: Yale University Press.

Frankie, G. W., Mata, A., & Vinson, S. B. (Eds.). (2004). *Biodiversity Conservation in Costa Rica: Learning the Lessons in a Seasonal Dry Forest.* Berkeley, USA: University of California Press.

Frankie, G. W., Thorp, R. W., Coville, R. E., & Ertter, B. (2014). *California Bees and Blooms: A Guide for Gardeners and Naturalists*. Berkeley, USA: Heyday.

Frankie, G. W., Thorp, R. W., Hernandez, J., Rizzardi, M., Ertter, B., Pawelek, J. C., Witt, S. L., Schindler, M., Coville, R., & Wojcik, V. A. (2009). Native bees are a rich natural resource in urban California gardens. *California Agriculture*, *63*(3), 113–120. https://doi.org/10.3733/ca.v063n03p113

Franzluebbers, A. J. (2005). Soil organic carbon sequestration and agricultural greenhouse gas emissions in the southeastern USA. *Soil and Tillage Research*, *83*(1), 120–147. http://doi.org/10.1016/j.still.2005.02.012

Fraser, J. A., Junqueira, A. B., Kawa, N. C., Moraes, C. P., & Clement, C. R. (2011). Crop diversity on anthropogenic dark earths in central Amazonia. *Human Ecology*, *39*(4), 395–406.

Fraser, R. H., Lantz, T. C., Olthof, I., Kokelj, S. V., & Sims, R. A. (2014). Warming-induced shrub expansion and lichen decline in the Western Canadian Arctic. *Ecosystems*, *17*(7), 1151–1168.

Freitas, L., Salino, A., Menini Neto, L., Almeida, T. E., Mortara, S. R., Stehmann, J. R., Amorim, A. M., Guimaraes, E. F., Coelho, M. N., Zanin, A., & Forzza, R. C. (2016). A comprehensive checklist of vascular epiphytes of the Atlantic Forest reveals outstanding endemic rates. *PhytoKeys*, *58*, 65–79. https://doi.org/10.3897/phytokeys.58.5643

Frelich, L. E. (1995). Old forest in the Lake States today and before European settlement. *Natural Areas Journal*, *15*(2), 157–167.

Frelich, L. E., Hale, C. M., Scheu, S., Holdsworth, A. R., Heneghan, L., Bohlen, P. J., & Reich, P. B. (2006). Earthworm invasion into previously earthworm-free temperate and boreal forests. *Biological Invasions*, *8*(2066), 1235–1245. http://doi.org/10.1007/s10530-006-9019-3

Frelich, L. E., & Reich, P. B. (2009). Wilderness conservation in an era of global warming and invasive species: a case study from Minnesota's Boundary Waters Canoe Area Wilderness. *Natural Areas Journal*, *29*(4), 385–393.

Frick, W. F., Cheng, T. L., Langwig, K. E., Hoyt, J. R., Janicki, A. F., Parise, K. L., Foster, J. T., & Kilpatrick, A. M. (2017). Pathogen dynamics during invasion and establishment of white-nose syndrome explain mechanisms of host persistence. *Ecology*, *98*(3), 624–631. https://doi.org/10.1002/ecy.1706/suppinfo

Fritz, S. A., Bininda-Emonds, O. R. P., & Purvis, A. (2009). Geographical variation in predictors of mammalian extinction risk: big is bad, but only in the tropics. *Ecology Letters*, *12*(6), 538–549.

Fuentes, N., Saldaña, A., Kühn, I., & Klotz, S. (2015). Climatic and socio-economic factors determine the level of invasion by alien plants in Chile. *Plant Ecology and Diversity*, *8*(3), 371–377. http://doi.org/10.1080/17550874.2014.984003

Fuller, P., & Neilson, M. (2015). The U.S. Geological Survey's Nonindigenous Aquatic Species Database: over thirty years of tracking introduced aquatic species in the United States (and counting).

Management of Biological Invasions, *6*(2), 159–170. http://doi.org/10.3391/mbi.2015.6.2.06

Gaines, S. D., & Lubchenco, J. (1982). A unified approach to marine plant-herbivore interactions II. Biogeography. *Annual Review of Ecology and Systematics*, *13*(1), 111–138. http://doi.org/10.1146/annurev.es.13.110182.000551

Galinier, J. (1997). Pueblos de la Sierra Madre. México: Centro de Estudios Mexicanos y Centroamericanos, Instituto Nacional Indigenista.

Galloway, J. N., Dentener, F. J., Capone, D. G., Boyer, E. W., Howarth, R. W., Seitzinger, S. P., Asner, G. P., Cleveland, C. C., Green, P. A., Holland, E. A., Karl, D. M., Michaels, A. F., Porter, J. H., Townsend, A. R., & Vörösmarty, C. J. (2004). Nitrogen cycles: past, present, and future. *Biogeochemistry*, *70*(2), 153–226. https://doi.org/10.1007/s10533-004-0370-0

Galloway, J. N., Townsend, A. R., Erisman, J. W., Bekunda, M., Cai, Z., Freney, J. R., Martinelli, L. A., Seitzinger, S. P., & Sutton, M. A. (2008). Transformation of the nitrogen cycle: recent trends, questions, and potential solutions. *Science*, *320*(5878), 889–892. https://doi.org/10.1126/science.1136674

Galluzzi, G., Eyzaguirre, P., & Negri, V. (2010). Home gardens: Neglected hotspots of agro-biodiversity and cultural diversity. *Biodiversity and Conservation*, *19*(13), 3635–3654. http://doi.org/10.1007/s10531-010-9919-5

Galluzzi, G., & López Noriega, I. (2014). Conservation and use of genetic resources of underutilized crops in the Americas—a continental analysis. *Sustainability*, *6*(2), 980–1017.

Gamfeldt, L., Snäll, T., Bagchi, R., Jonsson, M., Gustafsson, L., Kjellander, P., Ruiz-Jaen, M. C., Fröberg, M., Stendahl, J., Philipson, C. D., Mikusiński, G., Andersson, E., Westerlund, B., Andrén, H., Moberg, F., Moen, J., & Bengtsson, J. (2013). Higher levels of multiple ecosystem services are found in forests with more tree species. *Nature Communications*, *4*, 1340. https://doi.org/10.1038/ncomms2328

García-Guzman, P., Loayza, A. P., Carvajal, D. E., Letelier, L., & Squeo, F. A. (2012). The ecology, distribution and conservation status of *Myrcianthes coquimbensis* : a globally endangered endemic shrub of the Chilean Coastal Desert. *Plant Ecology and Diversity*, *5*(2), 197–204. http://doi.org/10.1080/17550874.2011.583286

García-Morales, R., Badano, E. I., & Moreno, C. E. (2013). Response of neotropical bat assemblages to human land use. *Conservation Biology*, *27*(5), 1096–1106. http://doi.org/10.1111/cobi.12099

García-Préchac, F., Ernst, O., Siri-Prieto, G., & Terra, J. A. (2004). Integrating no-till into crop-pasture rotations in Uruguay. *Soil and Tillage Research*, *77*(1), 1–13. http://doi.org/10.1016/j.still.2003.12.002

García, R. A., Engler, M. L., Peña, E., Pollnac, F. W., & Pauchard., A. (2015). Fuel characteristics of the invasive shrub *Teline monspessulana* (L.) K. Koch. *International Journal of Wildland Fire*, *24*(3), 372–379.

Gardner, T. A., Côté, I. M., Gill, J. A., Grant, A., & Watkinson, A. R. (2003). Long-term region-wide declines in Caribbean corals. *Science*, *301*(5635), 958–960. http://doi.org/10.1126/science.1086050

Garibaldi, L. A., Carvalheiro, L. G., Leonhardt, S. D., Aizen, M. A., Blaauw, B. R., Isaacs, R., Kuhlmann, M., Kleijn, D., Klein, A. M., Kremen, C., Morandin, L., Scheper, J., & Winfree, R. (2014). From research to action: enhancing crop yield through wild pollinators. *Frontiers in Ecology and the Environment*, *12*(8), 439–447. https://doi.org/10.1890/130330

Garitano-Zavala, Á., & Gismondi, P. (2003). Variación de la riqueza y diversidad de la ornitofauna en áreas verdes urbanas de las ciudades de La Paz y El Alto (Bolivia). *Ecología En Bolivia*, *38*(1), 65–78.

Gärtner, E., Rojas, G., & Castro, S. A. (2015). Compositional patterns of ruderal herbs in Santiago, Chile. *Gayana Botánica*, *72*(2), 192–202.

Gasparri, N. I., & Grau, H. R. (2009). Deforestation and fragmentation of Chaco dry forest in NW Argentina (1972-2007). *Forest Ecology and Management*, *258*(6), 913–921. http://doi.org/10.1016/j.foreco.2009.02.024

Gedan, K. B., Silliman, B. R., & Bertness, M. D. (2009). Centuries of human-driven change in salt marsh ecosystems. *Annual Review of Marine Science*, *1*(1), 117–141. http://doi.org/10.1146/annurev.marine.010908.163930

Germaine, S. S., & Wakeling, B. F. (2001). Lizard species distributions and habitat occupation along an urban gradient in Tucson, Arizona, USA. *Biological Conservation*, *97*(2), 229–237.

http://doi.org/10.1016/S0006-3207(00)00115-4

Germino, M. J., Chambers, J. C., & Brown, C. S. (2016). Introduction: Exotic annual *Bromus* in the Western USA. In M. J. Germino, J. C. Chambers, & C. S. Brown (Eds.), *Exotic Brome-Grasses in Arid and Semiarid Ecosystems of the Western US: Causes, Consequences, and Management Implications.* (pp. 1–7). New York, USA: Springer.

Geslin, B., & Morales, C. L. (2015). New records reveal rapid geographic expansion of *Bombus terrestris* Linnaeus, 1758 (Hymenoptera: Apidae), an invasive species in Argentina. *Check List*, *11*(3), 1620. http://doi.org/10.15560/11.3.1620

Gibbs, H. K., Ruesch, A. S., Achard, F., Clayton, M. K., Holmgren, P., Ramankutty, N., & Foley, J. A. (2010). Tropical forests were the primary sources of new agricultural land in the 1980s and 1990s. *PNAS*, *107*(38), 16732–16737. http://doi.org/10.1073/pnas.0910275107

Gill, F., & Donsker, D. (Eds.). (2017). *IOC World Bird List (v 7.3)*. Retrieved from www.worldbirdnames.org

Gilman, E. L., Ellison, J., Duke, N. C., & Field, C. (2008). Threats to mangroves from climate change and adaptation options. *Aquatic Botany*, *89*(2), 237–250.

Gleason, H. A. (1913). The relation of forest distribution and prairie fires in the Middle West. *Torreya*, *13*(8), 173–181. Retrieved from http://www.jstor.org/stable/40595396%5Cnhttp://about.jstor.org/terms

Global Forest Watch. (2017). Retrieved from http://www.globalforestwatch.org/map

Goettsch, B., & Hernández, H. M. (2006). Beta diversity and similarity among cactus assemblages in the Chihuahuan Desert. *Journal of Arid Environments*, *65*(4), 513–528. http://doi.org/10.1016/j.jaridenv.2005.08.008

Goettsch, B., Hilton-Taylor, C., Cruz-Piñón, G., Duffy, J. P., Frances, A., Hernández, H. M., Inger, R., Pollock, C., Schipper, J., Superina, M., Taylor, N. P., Tognelli, M., Abba, A. M., Arias, S., Arreola-Nava, H. J., Baker, M. A., Bárcenas, R. T., Barrios, D., Braun, P., Butterworth, C. A., Búrquez, A., Caceres, F., Chazaro-Basañez, M., Corral-Díaz, R., del Valle Perea, M., Demaio, P. H., Duarte de Barros, W. A., Durán, R., Faúndez Yancas, L., Felger, R. S., Fitz-Maurice, B., Fitz-Maurice, W. A., Gann, G., Gómez-Hinostrosa, C., Gonzales-Torres, L. R., Griffith, M. P., Guerrero, P. C., Hammel, B., Heil, K. D., Guadalupe Hernández-Oria, J., Hoffmann, M., Ishihara, M. I., Kiesling, R., Larocca, J., Luis León-de la Luz, J., Loaiza S., C. R., Lowry, M., Machado, M. C., Majure, L. C., Martínez Ávalos, J. G., Martorell, C., Maschinski, J., Méndez, E., Mittermeier, R. A., Nassar, J. M., Negrón-Ortiz, V., Oakley, L. J., Ortega-Baes, P., Ferreira, A. B. P., Pinkava, D. J., Porter, J. M., Puente-Martinez, R., Roque Gamarra, J., Saldivia Pérez, P., Sánchez Martínez, E., Smith, M., Sotomayor M. del C., J. M., Stuart, S. N., Tapia Muñoz, J. L., Terrazas, T., Terry, M., Trevisson, M., Valverde, T., Van Devender, T. R., Véliz-Pérez, M. E., Walter, H. E., Wyatt, S. A., Zappi, D., Alejandro Zavala-Hurtado, J., & Gaston, K. J. (2015). High proportion of cactus species threatened with extinction. *Nature Plants*, *1*(10), 15142. https://doi.org/10.1038/nplants.2015.142

Goijman, A. P., Conroy, M. J., Bernardos, J. N., & Zaccagnini, M. E. (2015). Multi-season regional analysis of multi-species occupancy: implications for bird conservation in agricultural lands in east-central Argentina. *PLoS ONE*, *10*(6), e0130874. http://doi.org/10.1371/journal.pone.0130874

Gómez-González, S., & Cavieres, L. A. (2009). Litter burning does not equally affect seedling emergence of native and alien species of the Mediterranean-type Chilean matorral. *International Journal of Wildland Fire*, *18*(2), 213–221. http://doi.org/10.1071/WF07074

Gómez-González, S., Torres-Díaz, C., Valencia, G., Torres-Morales, P., Cavieres, L. A., & Pausas, J. G. (2011). Anthropogenic fires increase alien and native annual species in the Chilean coastal matorral. *Diversity and Distributions*, *17*(1), 58–67. http://doi.org/10.1111/j.1472-4642.2010.00728.x

Gómez, M. F., Moreno, L. A., Andrade, G. G., & Rueda, C. (Eds.). (2016). *Biodiversidad 2015. Estado y Tendencias de la Biodiversidad Continental de Colombia*. Bogotá, D.C., Colombia: Instituto Alexander von Humboldt.

Gonçalves, F., Fischer, E., & Dirzo, R. (2017). Forest conversion to cattle ranching differentially affects taxonomic and functional groups of Neotropical bats. *Biological Conservation*, *210*, 343–348. http://doi.org/10.1016/j.biocon.2017.04.021

Gonçalves, G. S., Andrade, L. A. D., Xavier, K. R. F., & Silva, J. F. D. (2015). Control methods for *Prosopis juliflora* (Sw.) D.C. (Fabaceae) in invaded areas in the semiarid region of Brazil. *Ciência Florestal*, *25*(3), 645–653. http://doi.org/10.5902/1980509819615

González-Urrutia, M. (2009). Avifauna urbana de América Latina: estudio de casos. *Gestión Ambiental*, *17*, 55–68.

González Martínez, A. I., Barrios Caballero, Y., & De Jesus, S. (2017). Análisis de riesgo para especies invasoras en México. In *El Impacto de las Especies Exóticas en México* (pp. 24–29). Mexico City, Mexico: Centro de Estudios Sociales y de Opinión Pública de la Cámara de Diputados. Retrieved from https://www.diputados.gob.mx/cesop

González Oreja, J. A., Bonache Regidor, C., Buzo Franco, D., De La Fuente Díaz Ordaz, A., & Hernández Satín, L. (2007). Caracterización ecológica de la avifauna de los parques urbanos de la ciudad de Puebla (México). *Ardeola*, *54*(1), 53–67.

Goudie, A., & Seely, M. (2011). *World Heritage Desert Landscapes: Potential Priorities for the Recognition of Desert Landscapes and Geomorphological Sites on the World Heritage List*. Gland, Switzerland: International Union for Conservation of Nature (IUCN).

Gould, W. A., Martinuzzi, S., & Parés-Ramos, I. K. (2012). Land use, population dynamics, and land-cover change in eastern Puerto Rico. In S. F. Murphy & R. F. Stallard (Eds.), *Water Quality and Landscape Processes of Four Watersheds in Eastern Puerto Rico* (pp. 25–42). Reston, USA: U.S. Geological Survey / Professional Paper 1789.

Goulson, D., Nicholls, E., Botías, C., & Rotheray, E. L. (2015). Bee declines driven by combined stress from parasites, pesticides, and lack of flowers. *Science*, *347*(6229), 1255957. http://doi.org/10.1126/science.1255957

Government of Canada. (1991). *The Federal Policy on Wetland Conservation*. Ottawa, Canada. Retrieved from http://publications.gc.ca/collections/Collection/CW66-116-1991E.pdf

Government of Canada. (2009). *Canada's Fourth National Report to the United Nations Convention on Biological Diversity*. Retrieved from https://www.cbd.int/doc/world/ca/ca-nr-04-en.pdf

Gowda, J. H., Kitzberger, T., & Premoli, A. C. (2012). Landscape responses to a century of land use along the northern Patagonian forest-steppe transition. *Plant Ecology*, *213*(2), 259–272. http://doi.org/10.1007/s11258-011-9972-5

Grace, J. B., Anderson, T. M., Seabloom, E. W., Borer, E. T., Adler, P. B., Harpole, W. S., Hautier, Y., Hillebrand, H., Lind, E. M., Pärtel, M., Bakker, J. D., Buckley, Y. M., Crawley, M. J., Damschen, E. I., Davies, K. F., Fay, P. A., Firn, J., Gruner, D. S., Hector, A., Knops, J. M. H., MacDougall, A. S., Melbourne, B. A., Morgan, J. W., Orrock, J. L., Prober, S. M., & Smith, M. D. (2016). Integrative modelling reveals mechanisms linking productivity and plant species richness. *Nature*, *529*(7586), 390–393. https://doi.org/10.1038/nature16524

Gradinger, R., Bluhm, B. A., Hopcroft, R. R., Gebruk, A. V., Kosobokova, K., Sirenko, B., & Węsławski, J. M. (2010). Marine life in the Arctic. In A. D. McIntyre (Ed.), *Life in the World's Oceans: Diversity, Distribution, and Abundance* (pp. 183–202). New Jersey, USA: Willey Blackwell Publishing.

Grau, H. R., Gasparri, N. I., & Aide, T. M. (2005). Agriculture expansion and deforestation in seasonally dry forests of north-west Argentina. *Environmental Conservation*, *32*(2), 140–148.

Grau, H. R., Pérez, M., Martinuzzi, S., Encarnación, X., & Aide, T. M. (2008). Cambios socioeconómicos y regeneración del bosque en la República Dominicana. In M. González-Espinosa, J. M. Rey-Benayas, & N. Ramírez-Marcial (Eds.), *Restauración de Bosques en América Latina* (pp. 211–227). Ciudad de México, México: Fundación Internacional para la Restauración de Ecosistemas (FIRE) and Editorial Mundi-Prensa México.

Grez, A., Zaviezo, T., Gonzalez, G., & Rothman, S. (2010). *Harmonia axyridis* in Chile: a new threat. *Ciencia E Investigacion Agraria*, *37*(3), 145–149. http://doi.org/10.4067/S0718-16202010000300013

Grez, A., Zaviezo, T., Roy, H. E., Brown, P. M. J., & Bizama, G. (2016). Rapid spread of *Harmonia axyridis* in Chile and its effects on local coccinellid biodiversity. *Diversity and Distributions*, *22*(9), 982–994. http://doi.org/10.1111/ddi.12455

Griffin, D., & Anchukaitis, K. J. (2014). How unusual is the 2012-2014 California drought? *Geophysical Research Letters*, *41*(24), 9017–9023. http://doi.org/10.1002/2014GL062433

Grimm, N. B., Cook, E. M., Hale, R. L., Iwaniec, D. M., Seto, K. C., Solecki, W., & Griffith, C. A. (2015). A broader framing of ecosystem services in cities: benefits and challenges of built, natural, or hybrid system function. In K. C. Seto, W. D. Solecki, & C. A. Griffith (Eds.), *The Routledge Handbook of*

Urbanization and Global Environmental Change (pp. 203–212). New York, USA: Routeledge Taylor & Francis Group. Retrieved from https://www.routledgehandbooks.com/doi/10.4324/9781315849256.ch14

Grimm, N. B., Faeth, S. H., Golubiewski, N. E., Redman, C. L., Wu, J., Bai, X., & Briggs, J. M. (2008). Global change and the ecology of cities. *Science*, *319*(5864), 756–760. http://doi.org/10.1126/science.1150195

Groffman, P. M., Bain, D. J., Band, L. E., Belt, K. T., Brush, G. S., Grove, J. M., Pouyat, R. V., Yesilonis, I. C., & Zipperer, W. C. (2003). Down by the riverside: urban riparian ecology. *Frontiers in Ecology and the Environment*, *1*(6), 315–321. https://doi.org/10.2307/3868092

Groffman, P. M., Cavender-Bares, J., Bettez, N. D., Grove, J. M., Hall, S. J., Heffernan, J. B., Hobbie, S. E., Larson, K. L., Morse, J. L., Neill, C., Nelson, K., O'Neil-Dunne, J., Ogden, L., Pataki, D. E., Polsky, C., Chowdhury, R. R., & Steele, M. K. (2014). Ecological homogenization of urban USA. *Frontiers in Ecology and the Environment*, *12*(1), 74–81. https://doi.org/10.1890/120374

Gudynas, E. (2009). *El Mandato Ecológico. Derechos de la Naturaleza y Políticas Ambientales en la Nueva Constitución*. Quito, Ecuador: Abya Yala.

Gudynas, E. (2011). Buen vivir: germinando alternativas al desarrollo. *Revista América Latina En Movimiento*, *462*, 1–20.

Guerrero, P. C., Durán, A. P., & Walter, H. E. (2011). Latitudinal and altitudinal patterns of the endemic cacti from the Atacama desert to Mediterranean Chile. *Journal of Arid Environments*, *75*(11), 991–997. http://doi.org/10.1016/j.jaridenv.2011.04.036

Guido, A., & Pillar, V. D. (2017). Invasive plant removal: assessing community impact and recovery from invasion. *Journal of Applied Ecology*, *54*(4), 1230–1237. http://doi.org/10.1111/1365-2664.12848

Güneralp, B., & Seto, K. C. (2013). Futures of global urban expansion: uncertainties and implications for biodiversity conservation. *Environmental Research Letters*, *8*(1), 14025. http://doi.org/10.1088/1748-9326/8/1/014025

Guns, M., & Vanacker, V. (2013). Forest cover change trajectories and their impact on landslide occurrence in the tropical Andes. *Environmental Earth Sciences*, *70*(7), 2941–2952. http://doi.org/10.1007/s12665-013-2352-9

Guo, L. B., & Gifford, R. M. (2002). Soil carbon stocks and land use change: a meta analysis. *Global Change Biology*, *8*(4), 345–360. http://doi.org/10.1046/j.1354-1013.2002.00486.x

Gutiérrez, E. E., & Marinho-Filho, J. (2017). The mammalian faunas endemic to the Cerrado and the Caatinga. *ZooKeys*, (644), 105–157. http://doi.org/10.3897/zookeys.644.10827

Gutiérrez, R., & Schafleitner, R. (2007). *Caracterización Morfofisiológica, Molecular y de Procesamiento para Cultivares de Papas Nativas en la Provincia de Canchis, Cusco*. *Reporte de Investigación para Soluciones Prácticas-ITDG y Centro Internacional de la Papa*. Lima, Perú.

Haag, W. R. (2009). Past and future patterns of freshwater mussel extinctions in North America during the Holocene. In S. Turvey (Ed.), *Holocene Extinctions* (pp. 107–128). Oxford, UK: Oxford University Press.

Haberl, H., Erb, K. H., Krausmann, F., Gaube, V., Bondeau, A., Plutzar, C., Gingrich, S., Lucht, W., & Fischer-Kowalski, M. (2007). Quantifying and mapping the human appropriation of net primary production in earth's terrestrial ecosystems. *PNAS*, *104*(31), 12942–12947. https://doi.org/10.1073/pnas.0704243104

Haddad, N. M., Brudvig, L. A., Clobert, J., Davies, K. F., Gonzalez, A., Holt, R. D., Lovejoy, T. E., Sexton, J. O., Austin, M. P., Collins, C. D., Cook, W. M., Damschen, E. I., Ewers, R. M., Foster, B. L., Jenkins, C. N., King, A. J., Laurance, W. F., Levey, D. J., Margules, C. R., Melbourne, B. A., Nicholls, A. O., Orrock, J. L., Song, D.-X., & Townshend, J. R. (2015). Habitat fragmentation and its lasting impact on Earth's ecosystems. *Science Advances*, *1*(2), e1500052.https://doi.org/10.1126/sciadv.1500052

Hale, C. M., Frelich, L. E., Reich, P. B., & Pastor, J. (2005). Effects of European earthworm invasion on soil characteristics in northern hardwood forests of Minnesota, USA. *Ecosystems*, *8*(8), 911–927. http://doi.org/10.1007/s10021-005-0066-x

Hall, S. J., Learned, J., Ruddell, B., Larson, K. L., Cavender-Bares, J., Bettez, N., Groffman, P. M., Grove, J. M., Heffernan, J. B., Hobbie, S. E., Morse, J. L., Neill, C., Nelson, K. C., ONeil-Dunne, J. P. M., Ogden, L., Pataki, D. E., Pearse, W. D., Polsky, C., Chowdhury, R. R., Steele, M. K., & Trammell, T. L. E. (2016).

Convergence of microclimate in residential landscapes across diverse cities in the United States. *Landscape Ecology*, *31*(1), 101–117. https://doi.org/10.1007/s10980-015-0297-y

Halpern, B. S., Walbridge, S., Selkoe, K. A., Kappel, C. V., Micheli, F., D'Agrosa, C., Bruno, J. F., Casey, K. S., Ebert, C., Fox, H. E., Fujita, R., Heinemann, D., Lenihan, H. S., Madin, E. M. P., Perry, M. T., Selig, E. R., Spaldin, M., Steneck, R., & Watson, R. (2008). A global map of human impact on marine ecosystems. *Science*, *319*(5865), 948–952. https://doi.org/10.1126/science.1149345

Halsey, R., & Syphard, A. (2016). High-severity fire in chaparral: cognitive dissonance in the shrublands. In D. A. DellaSala & C. T. Hanson (Eds.), *The Ecological Importance of Mixed-Severity Fires: Nature's Phoenix* (p. 177–209.). New York, USA: Elsevier.

Hamer, A. J., & McDonnell, M. J. (2008). Amphibian ecology and conservation in the urbanising world: A review. *Biological Conservation*, *141*(10), 2432–2449. http://doi.org/10.1016/j.biocon.2008.07.020

Hammel, B. (1990). The distribution of diversity among families, genera and habitat types in the La Selva Flora. In A. H. Gentry (Ed.), *Four Neotropical Rainforests* (pp. 75–84). New Haven, USA: Yale University Press.

Hansen, M. C., Potapov, P. V., Moore, R., Hancher, M., Turubanova, S. A., Tyukavina, A., Thau, D., Stehman, S. V., Goetz, S. J., Loveland, T. R., Kommareddy, A., Egorov, A., Chini, L., Justice, C. O., & Townshend, J. R. G. (2013). High-resolution global maps of 21st-century forest cover change. *Science*, *342*(6160), 850–853. https://doi.org/10.1126/science.1244693

Harlan, J. (1961). Geographic origin of plants useful to agriculture. In *Germplasm Resources* (pp. 3–19). Washington D.C., USA: American Association for the Advancement of Science.

Harlan, J. R. (1971). Agricultural origins: centers and noncenters. *Science*, *174*(4008), 468–474. http://doi.org/10.1126/science.174.4008.468

Harris, J. B. C., Reid, J. L., Scheffers, B. R., Wanger, T. C., Sodhi, N. S., Fordham, D. A., & Brook, B. W. (2012). Conserving imperiled species: a comparison of the IUCN Red List and U.S. Endangered Species Act. *Conservation Letters*, *5*(1), 64–72.

Harsch, M. A., Hulme, P. E., McGlone, M. S., & Duncan, R. P. (2009). Are treelines advancing? A global meta-analysis of treeline response to climate warming. *Ecology Letters*, *12*(10), 1040–1049.

Harvell, C. D. (1999). Emerging marine diseases--climate links and anthropogenic factors. *Science*, *285*(5433), 1505–1510. http://doi.org/10.1126/science.285.5433.1505

Hatcher, M. J., Dick, J. T. A., & Dunn, A. M. (2012). Disease emergence and invasions. *Functional Ecology*, *26*(6), 1275–1287. http://doi.org/10.1111/j.1365-2435.2012.02031.x

Hautier, Y., Seabloom, E. W., Borer, E. T., Adler, P. B., Harpole, W. S., Hillebrand, H., Lind, E. M., MacDougall, A. S., Stevens, C. J., Bakker, J. D., Buckley, Y. M., Chu, C., Collins, S. L., Daleo, P., Damschen, E. I., Davies, K. F., Fay, P. A., Firn, J., Gruner, D. S., Jin, V. L., Klein, J. A., Knops, J. M. H., La Pierre, K. J., Li, W., McCulley, R. L., Melbourne, B. A., Moore, J. L., O'Halloran, L. R., Prober, S. M., Risch, A. C., Sankaran, M., Schuetz, M., & Hector, A. (2014). Eutrophication weakens stabilizing effects of diversity in natural grasslands. *Nature*, *508*(7497), 521–525. https://doi.org/10.1038/nature13014

Hayes, D. J., Kicklighter, D. W., McGuire, A. D., Chen, M., Zhuang, Q., Yuan, F., Melillo, J. M., & Wullschleger, S. D. (2014). The impacts of recent permafrost thaw on land-atmosphere greenhouse gas exchange. *Environmental Research Letters*, *9*(4), 045005. https://doi.org/10.1088/1748-9326/9/4/045005

Heath, L. S., Smith, J. E., Skog, K. E., Nowak, D. J., & Woodall, C. W. (2011). Managed forest carbon estimates for the US greenhouse gas inventory, 1990-2008. *Journal of Forestry*, *109*(3), 167–173.

Hector, A., Bell, T., Hautier, Y., Isbell, F., Kéry, M., Reich, P. B., van Ruijven, J., & Schmid, B. (2011). BUGS in the analysis of biodiversity experiments: species richness and composition are of similar importance for grassland productivity. *PLoS One*, *6*(3), e17434. https://doi.org/10.1371/journal.pone.0017434

Hendrix, P. F., Baker, G. H., Calaham, M. A., Damoff, G. A., Fragoso, C., Gonzalez, G., James, S. W., Lachnicht, S. L., Winsome, T., & Zou, X. (2006). Invasion of exotic earthworms into ecosystems inhabited by native earthworms. *Biological Invasions*, *8*(6), 1287–1300. https://doi.org/10.1007/978-1-4020-5429-7_9

Herms, D. A., & McCullough, D. G. (2014). Emerald ash borer invasion of North America: history, biology, ecology, impacts, and management. *Annual Review of Entomology*, *59*, 13–30. http://doi.org/10.1146/annurev-ento-011613-162051

Hernández, A., Miranda, M. D., Arellano, E. C., & Dobbs, C. (2016). Landscape trajectories and their effect on fragmentation for a Mediterranean semi-arid ecosystem in Central Chile. *Journal of Arid Environments*, *127*, 74–81.

Hernández, G. (Ed.). (1999). *Mesoamerican Wetlands. Ramsar sites in Central America and Mexico*. San José, Costa Rica: Unión Mundial para la Naturaleza (UICN)/ ORMA.

Hernández, X. (1985). Biología agrícola: los conocimientos biológicos y su aplicación a la agricultura. México: CECSA.

Hicke, J. A., Asner, G. P., Randerson, J. T., Tucker, C. J., Los, S., Birdsey, R. A., Jenkins, J. C., & Field, C. B. (2002). Trends in North American net primary productivity derived from satellite observations, 1982-1998. *Global Biogeochemical Cycles*, *16*(2), 1019. https://doi.org/10.1029/2001GB001550

Higgins, S. N., & Vander Zanden, M. J. (2010). What a difference a species makes: a meta-analysis of dreissenid mussel impacts on freshwater ecosystems. *Ecological Monographs*, *80*(2), 179–196.

Highly Migratory Species Management Division. (2006). *SEDAR 11: Stock Assessment Report: Large Coastal Shark Complex, Blacktip and Sandbar Shark*. Silver Spring, USA: National Marine Fisheries Service (NMFS). Retrieved from http://sedarweb.org/docs/sar/Final_LCS_SAR.pdf

Hijmans, R. J. (2016). Raster: Geographic Data Analysis and Modeling. R package version 2.5-8. Retrieved from https://cran.r-project.org/package=raster

Himsworth, C. G., Parsons, K. L., Jardine, C., & Patrick, D. M. (2013). Rats, cities, people, and pathogens: A systematic review and narrative synthesis of literature regarding the ecology of rat-associated zoonoses in urban centers. *Vector-Borne and Zoonotic Diseases*, *13*(6), 349–359. http://doi.org/10.1089/vbz.2012.1195

Hinkel, K. M., Eisner, W. R., & Kim, C. J. (2017). Detection of tundra trail damage near Barrow, Alaska using remote imagery. *Geomorphology*, *293*, 360–367. Retrieved from https://doi.org/10.1016/j.geomorph.2016.09.013

Hinzman, L. D., Bettez, N. D., Bolton, W. R., Chapin, F. S., Dyurgerov, M. B., Fastie, C. L., Griffith, B., Hollister, R. D., Hope, A., Huntington, H. P., Jensen, A. M., Jia, G. J., Jorgenson, T., Kane, D. L., Klein, D. R., Kofinas, G., Lynch, A. H., Lloyd, A. H., McGuire, A. D., Nelson, F. E., Oechel, W. C., Osterkamp, T. E., Racine, C. H., Romanovsky, V. E., Stone, R. S., Stow, D. A., Sturm, M., Tweedie, C. E., Vourlitis, G. L., Walker, M. D., Walker, D. A., Webber, P. J., Welker, J. M., Winker, K. S., & Yoshikawa, K. (2005). Evidence and implications of recent climate change in northern Alaska and other Arctic regions. *Climatic Change*, *72*(3), 251–298. https://doi.org/10.1007/s10584-005-5352-2

Hobbs, R. J., & Mooney, H. A. (1998). 1998 report. Retrieved from http://www.dfg.ca.gov/wildlife/nongame/t_e_spp/

Hoekstra, J. M., Boucher, T. M., Ricketts, T. H., & Roberts, C. (2005). Confronting a biome crisis: global disparities of habitat loss and protection. *Ecology Letters*, *8*(1), 23–29. http://doi.org/10.1111/j.1461-0248.2004.00686.x

Hoekstra, J. M., Molnar, J. L., Jennings, M., Revenga, C., Spalding, M. D., Boucher, T. M., Robertson, J. C., Heibel, T. J., & Elllison, K. (2010). *The Atlas of Global Conservation: Changes, Challenges, and Opportunities to Make a Difference*. (J. L. Molnar, Ed.). Berkeley, USA: University of California Press.

Hofstede, R., Calles, J., López, V., Polanco, R., Torres, F., Ulloa, J., Vásquez, A., & Cerra, M. (2014). *Los Páramos Andinos ¿Qué Sabemos? Estado de Conocimiento sobre el Impacto del Cambio Climático en el Ecosistema Páramo*. Quito, Ecuador: UICN. Retrieved from https://portals.iucn.org/library/sites/library/files/documents/2014-025.pdf

Hofstede, R. G. M., Groenendijk, J. P., Coppus, R., Fehse, J. C., & Sevink, J. (2002). Impact of pine plantations on soils and vegetation in the Ecuadorian high Andes. *Mountain Research and Development*, *22*(2), 159–167. http://doi.org/10.1659/0276-4741(2002)022%5B0159:IOPPOS%5D2.0.CO;2

Holdsworth, A. R., Frelich, L. E., & Reich, P. B. (2007). Effects of earthworm invasion on plant species richness in northern hardwood forests. *Conservation Biology*, *21*(4), 997–1008. http://doi.org/10.1111/j.1523-1739.2007.00740.x

Holmlund, C. M., & Hammer, M. (1999). Ecosystem services generated by fish populations. *Ecological Economics*, *29*(2), 253–268. http://doi.org/10.1016/S0921-8009(99)00015-4

Hooper, D. U., Adair, E. C., Cardinale, B. J., Byrnes, J. E. K., Hungate, B. A., Matulich, K. L., Gonzalez, A., Duffy, J. E., Gamfeldt, L., & O'Connor, M. I. (2012). A global synthesis reveals biodiversity loss as a major driver of ecosystem change. *Nature*, *486*(7401), 105–108. https://doi.org/10.1038/nature11118

Hope, D., Gries, C., Zhu, W., Fagan, W. F., Redman, C. L., Grim, N. B., Nelson, A. L., Martin, C., & Kinzig, A. (2003). Socioeconomics drive urban plant diversity. *PNAS*, *100*(15), 8788–8792. https://doi.org/10.1073/pnas.1537557100

Hourigan, T. F. (1999). Conserving ocean biodiversity: trends and challenges. In B. Cicin-Sain, R. W. Knecht, & N. Foster (Eds.), *Trends and Future Challenges for U.S. National Ocean and Coastal Policy*. Washington D.C., USA: Proceedings of a Workshop.

Houspanossian, J., Giménez, R., Jobbágy, E., & Nosetto, M. (2017). Surface albedo raise in the South American Chaco: combined effects of deforestation and agricultural changes. *Agricultural and Forest Meteorology*, *232*, 118–127. http://doi.org/10.1016/j.agrformet.2016.08.015

Houtart, F. (2014). El concepto de sumak kawsay (Buen vivir) y su correspondencia con el bien común de la humanidad. In G. C. Delgado (Ed.), *Buena Vida, Buen Vivir: Imaginarios Alternativos Para el Bien Común de la Humanidad* (pp. 97–123). México D.F., México: Universidad Nacional Autónoma de México.

Hoverman, J. T., Mihaljevic, J. R., Richgels, K. L. D., Kerby, J. L., & Johnson, P. T. J. (2012). Widespread co-occurrence of virulent pathogens within California amphibian communities. *EcoHealth*, *9*(3), 288–292. http://doi.org/10.1007/s10393-012-0778-2

Howard, B. C. (2016). Pablo Escobar's escaped hippos are thriving in Colombia. *National Geographic*. Retrieved from https://news.nationalgeographic.com/2016/05/160510-pablo-escobar-hippos-colombia/

Hoyos, L. E., Cingolani, A. M., Zak, M. R., Vaieretti, M. V., Gorla, D. E., & Cabido, M. R. (2013). Deforestation and precipitation patterns in the arid Chaco forests of central Argentina. *Applied Vegetation Science*, *16*(2), 260–271. http://doi.org/10.1111/j.1654-109X.2012.01218.x

Hribljan, J. A., Cooper, D. J., Sueltenfuss, J., Wolf, E. C., Heckman, K. A., Lilleskov, E. A., & Chimner, R. A. (2015). Carbon storage and long-term rate of accumulation in high-altitude Andean peatlands of Bolivia. *Mires and Peat*, *15*(12), 1–14. Retrieved from http://www.mires-and-peat.net/,

Hribljan, J. A., Suárez, E., Heckman, K. A., Lilleskov, E. A., & Chimner, R. A. (2016). Peatland carbon stocks and accumulation rates in the Ecuadorian páramo. *Wetlands Ecology and Management*, *24*(2), 113–127. http://doi.org/10.1007/s11273-016-9482-2

Huamantupa, I., Cuba, M., Urrunaga, R., Paz, E., Ananya, N., Callalli, M., Pallquil, N., & Coasaca, H. (2011). Richness, use and origin of expended medicinal plants in the markets of the Cusco City. *Revista Peruana de Biologia*, *18*(3), 283–291.

Hucke-Gaete, R., Crespo, E. ., & Schlatter, R. P. (Eds.). (2004). *Aquatic Mammals in Latin America: Proceedings of a Workshop on Identifying High-Priority Conservation Needs and Actions*. Bonn, Germany: UNEP/CMS Secretariat. Retrieved from http://www.iucn-csg.org/wp-content/uploads/2010/03/final-report-cms-latam-workshop.pdf

Hugelius, G., Tarnocai, C., Broll, G., Canadell, J. G., Kuhry, P., & Swanson, D. K. (2013). The northern circumpolar soil carbon database: Spatially distributed datasets of soil coverage and soil carbon storage in the northern permafrost regions. *Earth System Science Data*, *5*(1), 3–13. http://doi.org/10.5194/essd-5-3-2013

Hung, H., Kallenborn, R., Breivik, K., Su, Y., Brorström-Lundén, E., Olafsdottir, K., Thorlacius, J. M., Leppänen, S., Bossi, R., Skov, H., Manø, S., Patton, G. W., Stern, G., Sverko, E., & Fellin, P. (2010). Atmospheric monitoring of organic pollutants in the Arctic under the Arctic Monitoring and Assessment Programme (AMAP): 1993-2006. *Science of the Total Environment*, *408*(15), 2854–2873. https://doi.org/10.1016/j.scitotenv.2009.10.044

Hunke, P., Mueller, E. N., Schröder, B., & Zeilhofer, P. (2015). The Brazilian Cerrado: assessment of water and soil degradation in catchments under intensive agriculture use. *Ecohydrology*, *8*(6), 1154–1180.

Hutchings, J. A., Côté, I. M., Dodson, J. J., Fleming, I. A., Jennings, S., Mantua, N. J., Peterman, R. M., Riddell,

B. E., Weaver, A. J., & VanderZwaag, D. L. (2012). *Sustaining Canada's Marine Biodiversity: Responding to the Challenges Posed by Climate Change, Fisheries, and Aquaculture*. Ottawa, Canada: Royal Society of Canada. Retrieved from http://www.ianas.org/docs/books/wbp08.pdf

I3N. (2016). I3N National Database: Argentina, Brasil, Chile, Colombia, Costa Rica, Jamaica, and Uruguay. I3N National Database.

Ide, S. M., Ruiz, C. G., Sandoval, A. C., & Valenzuela, J. E. (2011). Detección de *Thaumastocoris peregrinus* (Hemiptera: Thaumastocoridae) asociado a *Eucalyptus spp.* en Chile. *Bosque*, *32*(3), 309–313. http://doi.org/10.4067/S0717-92002011000300012

Ingrao, D. A., Mikkelson, P. M., & Hicks, D. W. (2001). Another introduced marine mollusk in the Gulf of Mexico: the indo- Pacific green mussel, *Perna viridis*, in Tampa Bay, Florida. *Journal of Shellfish Research*, *20*(1), 13–19.

Instituto de Asuntos Públicos-Centro de Análisis de Políticas Públicas. (2016). *Informe País: Estado del Medio Ambiente en Chile. Comparación 1999 - 2015*. Santiago, Chile: Universidad de Chile. Retrieved from http://www.uchile.cl/publicaciones/129607/informe-pais-estado-del-medio-ambiente-en-chile-1999-2015

International Labour Organization. (1989). *Indigenous and Tribal Peoples Convention, C169*. International Labour Organization (ILO). Retrieved from http://www.refworld.org/docid/3ddb6d514.html

Iñiguez-Armijos, C., Leiva, A., Frede, H. G., Hampel, H., & Breuer, L. (2014). Deforestation and benthic indicators: how much vegetation cover is needed to sustain healthy Andean streams? *PLoS One*, *9*(8), e105869. http://doi.org/10.1371/journal.pone.0105869

Iriarte, J. A., Lobos, G. A., & Jaksic, F. M. (2005). Invasive vertebrate species in Chile and their control and monitoring by governmental agencies. *Revista Chilena de Historia Natural*, *78*(1), 143–154. http://doi.org/10.4067/s0716-078x2005000100010

Iroumé, A., & Palacios, H. (2013). Afforestation and changes in forest composition affect runoff in large river basins with pluvial regime and Mediterranean climate, Chile. *Journal of Hydrology*, *505*, 113–125. http://doi.org/10.1016/j.jhydrol.2013.09.031

Isaac, N. J. B., Turvey, S. T., Collen, B., Waterman, C., & Baillie, J. E. M. (2007). Mammals on the EDGE: conservation priorities based on threat and phylogeny. *PLoS ONE*, *2*(3), e296. http://doi.org/10.1371/journal.pone.0000296

Isbell, F. I., Polley, H. W., & Wilsey, B. J. (2009). Species interaction mechanisms maintain grassland plant species diversity. *Ecology*, *90*(7), 1821–1830. http://doi.org/10.1890/08-0514.1

Iturraspe, R. (2010). *Las turberas de Tierra del Fuego y el Cambio Climático Global*. Buenos Aires, Argentina: Fundación Humedales / Wetlands International.

IUCN. (2009). The IUCN Red list of Threatened Species Version 2009-1. Retrieved from http://www.iucnredlist.org/

IUCN. (2014). IUCN Red List of Threatened Species. Version 2014.2. Retrieved August 31, 2017, from http://www.iucnredlist.org

IUCN. (2017). The IUCN Red List of Threatened Species. Version 2016-3. Retrieved January 1, 2017, from http://www.iucnredlist.org

IUCN, & SSC PBSG. (2017). Summary of polar bear population status per 2017. Retrieved from http://pbsg.npolar.no/en/status/status-table.html

Jackson, J. B. C., Donovan, M. K., Cramer, K. L., & Lam, V. (Eds.). (2014). *Status and Trends of Caribbean Coral Reefs: 1970-2012*. Gland, Switzerland: Global Coral Reef Monitoring Network, IUCN.

Jackson, R. B., Banner, J. L., Jobbágy, E. G., Pockman, W. T., & Wall, D. H. (2002). Ecosystem carbon loss with woody plant invasion of grasslands. *Nature*, *418*(6898), 623–626. http://doi.org/10.1038/nature00910

Jackson, R. B., Jobbágy, E. G., & Avissar, R. (2005). Trading water for carbon with biological carbon sequestration. *Science*, *310*(5756), 1944–1947. http://doi.org/10.1126/science.1119282

Jackson, R. B., Lajtha, K., Crow, S. E., Hugelius, G., Kramer, M. G., & Piñeiro, G. (2017). The ecology of soil carbon: pools, vulnerabilities, and biotic and abiotic controls. *Annual Review of Ecology, Evolution, and Systematics*, *48*(1), 419–445. http://doi.org/10.1146/annurev-ecolsys-112414-054234

Jackson, R. B., Randerson, J. T., Canadell, J. G., Anderson, R. G., Avissar, R., Baldocchi, D. D., Bonan, G. B., Caldeira, K., Diffenbaugh, N. S., Field, C. B., Hungate, B. A., Jobbágy, E. G., Kueppers, L. M., Nosetto,

M. D., & Pataki, D. E. (2008). Protecting climate with forests. *Environmental Research Letters*, *3*(4), 044006. https://doi.org/10.1088/1748-9326/3/4/044006

Jaksic, F. M. (1998). Vertebrate invaders and their ecological impacts in Chile. *Biodiversity and Conservation*, *7*(11), 1427–1445. http://doi.org/10.1023/A:1008825802448

Jaksic, F. M., Pavez, E. F., Jiménez, J. E., & Torres-Mura, J. C. (2001). The conservation status of raptors in the Metropolitan Region, Chile. *Journal of Raptor Research*, *35*(2), 151–158.

Janzen, D. H. (1987). Insect diversity of a Costa Rican dry forest: why keep it, and how? *Biological Journal of the Linnean Society*, *30*(4), 343–356. http://doi.org/10.1111/j.1095-8312.1987.tb00307.x

Janzen, D. H. (1988). Tropical dry forests: the most endangered major tropical ecosystem. In E. O. Wilson & F. M. Peters (Eds.), *Biodiversity* (pp. 130–137). Washington, D.C.: National Academy Press.

Janzen, D. H. (2002). Tropical Dry Forest: Area de Conservación Guanacaste, northwetern Costa Rica. In M. R. Perrow & A. J. Davy (Eds.), *Handbook of Ecological Restoration. Vol 2 Restoration in Practice* (pp. 559–583). Cambridge, UK: Cambridge University Press.

Janzen, D. H., & Hallwachs, W. (2016). Biodiversity conservation history and future in Costa Rica: The case of Área de Conservación Guanacaste (ACG). In M. Kappelle (Ed.), *Costa Rican Ecosystems* (pp. 290–341). Chicago, USA.: University Of Chicago Press. http://doi.org/10.7208/chicago/9780226121642.003.0010

Jenkins, C. N., & Joppa, L. (2009). Expansion of the global terrestrial protected area system. *Biological Conservation*, *142*(10), 2166–2174. http://doi.org/10.1016/j.biocon.2009.04.016

Jetz, W., Cavender-Bares, J., Pavlick, R., Schimel, D., Davis, F. W., Asner, G. P., Guralnick, R., Kattge, J., Latimer, A. M., Moorcroft, P., Schaepman, M. E., Schildhauer, M. P., Schneider, F. D., Schrodt, F., Stahl, U., & Ustin, S. L. (2016). Monitoring plant functional diversity from space. *Nature Plants*, *2*(3), 16024. https://doi.org/10.1038/nplants.2016.24

Jetz, W., Thomas, G. H., Joy, J. B., Hartmann, K., & Mooers, A. O. (2012). The global diversity of birds in space and time. *Nature*, *491*(7424), 444–448.

Jetz, W., Thomas, G. H., Joy, J. B., Redding, D. W., Hartmann, K., & Mooers, A. O. (2014). Global distribution and conservation of evolutionary distinctness in birds. *Current Biology*, *24*(9), 919–930. http://doi.org/10.1016/j.cub.2014.03.011

Jiménez, A., Pauchard, A., Cavieres, L. A., Marticorena, A., & Bustamante, R. O. (2008). Do climatically similar regions contain similar alien floras? A comparison between the mediterranean areas of central Chile and California. *Journal of Biogeography*, *35*(4), 614–624. http://doi.org/10.1111/j.1365-2699.2007.01799.x

Jiménez, J. E., Crego, R. D., Soto, G. E., Roman, I., Rozzi, R., & Vergara, P. M. (2014). Potential impact of the alien American mink (*Neovison vison*) on Magellanic woodpeckers (Campephilus magellanicus) in Navarino Island, Southern Chile. *Biological Invasions*, *16*(4), 961–966. http://doi.org/10.1007/s10530-013-0549-1

Jiménez, M. A., Jaksic, F. M., Armesto, J. J., Gaxiola, A., Meserve, P. L., Kelt, D. A., & Gutiérrez, J. R. (2011). Extreme climatic events change the dynamics and invasibility of semi-arid annual plant communities. *Ecology Letters*, *14*(12), 1227–1235. http://doi.org/10.1111/j.1461-0248.2011.01693.x

Johnson, A. M., & Karels, T. J. (2016). Partitioning the effects of habitat fragmentation on rodent species richness in an urban landscape. *Urban Ecosystems*, *19*(2), 547–560. http://doi.org/10.1007/s11252-015-0513-1

Johnson, E. A., & Miyanishi, K. (2008). Creating new landscapes and ecosystems. *Annals of the New York Academy of Sciences*, *1134*(1), 120–145. http://doi.org/10.1196/annals.1439.007

Joly, C. A., Metzger, J. P., & Tabarelli, M. (2014). Experiences from the Brazilian Atlantic Forest: ecological findings and conservation initiatives. *New Phytologist*, *204*(3), 459–473.

Jones, K. E., Patel, N. G., Levy, M. A., Storeygard, A., Balk, D., Gittleman, J. L., & Daszak, P. (2008). Global trends in emerging infectious diseases. *Nature*, *451*(7181), 990–993. http://doi.org/10.1038/nature06536

Jørgensen, P. M., Nee, M. H., & Beck, S. G. (2015). Catálogo de plantas vasculares de Bolivia. In *Monographs in Systematic Botany from the Missouri Botanical Garden* (4th ed., Vol. 127). Missouri,USA: Missouri Botanical Garden Press.

Jose, S., Gold, M. A., & Garrett, H. E. (2012). The future of temperate agroforestry in the United States. In P. K. R. Nair & D. Garrity (Eds.), *Agroforestry - The Future of Global Land Use* (pp. 217–245). New York, USA: Springer.

Junk, W. J. (2007). Freshwater fishes of South America: their biodiversity, fisheries, and habitats- a synthesis. *Aquatic Ecosystem Health & Management*, *10*(2), 228–242.

Junk, W. J., Brown, M., Campbell, I. C., Finlayson, M., Gopal, B., Ramberg, L., & Warner, B. G. (2006). The comparative biodiversity of seven globally important wetlands: a synthesis. *Aquatic Sciences*, *68*(3), 400–414. http://doi.org/10.1007/s00027-006-0856-z

Junk, W. J., Piedade, M. T. F., Lourival, R., Wittmann, F., Kandus, P., Lacerda, L. D., Bozelli, R. L., Esteves, F. A., Nunes da Cunha, C., Maltchik, L., Schöngart, J., Schaeffer-Novelli, Y., & Agostinho, A. A. (2014). Brazilian wetlands: their definition, delineation, and classification for research, sustainable management, and protection. *Aquatic Conservation: Marine and Freshwater Ecosystems*, *24*(1), 5–22. https://doi.org/10.1002/aqc.2386

Junk, W. J., Soares, M. G. M., & Bayley, P. B. (2007). Freshwater fishes of the Amazon River basin: their biodiversity, fisheries, and habitats. *Aquatic Ecosystem Health & Management*, *10*(2), 153–173. http://doi.org/10.1080/14634980701351023

Juri, M. D., & Chani, J. M. (2009). Variación estacional en la composición de las comunidades de aves en un gradiente urbano. *Ecología Austral*, *19*(3), 175–184.

Juzwik, J., Appel, D. N., MacDonald, W. L., & Burks, S. (2011). Challenges and successes in managing oak wilt in the United States. *Plant Disease*, *95*(8), 888–900. http://doi.org/10.1094/PDIS-12-10-0944

Kairo, M., Ali, B., Cheesman, O., Haysom, K., & Murphy, S. (2003). *Invasive Species Threats in the Caribbean Region, Report to the Nature Conservancy*. Arlington, USA: CAB International.

Kane, V. R., Cansler, C. A., Povak, N. A., Kane, J. T., McGaughey, R. J., Lutz, J. A., Churchill, D. J., & North, M. P. (2015). Mixed severity fire effects within the Rim fire: relative importance of local climate, fire weather, topography, and forest structure. *Forest Ecology and Management*, *358*, 62–79. https://doi.org/10.1016/j.foreco.2015.09.001

Kattge, J., Díaz, S., Lavorel, S., Prentice, I. C., Leadley, P., Bönisch, G., Garnier, E., Westoby, M., Reich, P. B., Wright, I. J., Cornelissen, J. H. C., Violle, C., Harrison, S. P., Van Bodegom, P. M., Reichstein, M., Enquist, B. J., Soudzilovskaia, N. A., Ackerly, D. D., Anand, M., Atkin, O., Bahn, M., Baker, T. R., Baldocchi, D., Bekker, R., Blanco, C. C., Blonder, B., Bond, W. J., Bradstock, R., Bunker, D. E., Casanoves, F., Cavender-Bares, J., Chambers, J. Q., Chapin, F. S., Chave, J., Coomes, D., Cornwell, W. K., Craine, J. M., Dobrin, B. H., Duarte, L., Durka, W., Elser, J., Esser, G., Estiarte, M., Fagan, W. F., Fang, J., Fernández-Méndez, F., Fidelis, A., Finegan, B., Flores, O., Ford, H., Frank, D., Freschet, G. T., Fyllas, N. M., Gallagher, R. V., Green, W. A., Gutierrez, A. G., Hickler, T., Higgins, S. I., Hodgson, J. G., Jalili, A., Jansen, S., Joly, C. A., Kerkhoff, A. J., Kirkup, D., Kitajima, K., Kleyer, M., Klotz, S., Knops, J. M. H., Kramer, K., Kühn, I., Kurokawa, H., Laughlin, D., Lee, T. D., Leishman, M., Lens, F., Lenz, T., Lewis, S. L., Lloyd, J., Llusià, J., Louault, F., Ma, S., Mahecha, M. D., Manning, P., Massad, T., Medlyn, B. E., Messier, J., Moles, A. T., Müller, S. C., Nadrowski, K., Naeem, S., Niinemets, Ü., Nöllert, S., Nüske, A., Ogaya, R., Oleksyn, J., Onipchenko, V. G., Onoda, Y., Ordoñez, J., Overbeck, G., Ozinga, W. A., Patiño, S., Paula, S., Pausas, J. G., Peñuelas, J., Phillips, O. L., Pillar, V., Poorter, H., Poorter, L., Poschlod, P., Prinzing, A., Proulx, R., Rammig, A., Reinsch, S., Reu, B., Sack, L., Salgado-Negret, B., Sardans, J., Shiodera, S., Shipley, B., Siefert, A., Sosinski, E., Soussana, J. F., Swaine, E., Swenson, N., Thompson, K., Thornton, P., Waldram, M., Weiher, E., White, M., White, S., Wright, S. J., Yguel, B., Zaehle, S., Zanne, A. E., & Wirth, C. (2011). TRY- a global database of plant traits. *Global Change Biology*, *17*(9), 2905–2935. https://doi.org/10.1111/j.1365-2486.2011.02451.x

Keddy, P. A., Fraser, L. H., Solomeshch, A. I., Junk, W. J., Campbell, D. R., Arroyo, M. T. K., & Alho, C. J. R. (2009). Wet and wonderful: the world’s largest wetlands are conservation priorities. *BioScience*, *59*(1), 39–51. http://doi.org/10.1525/bio.2009.59.1.8

Keeley, J. E., & Zedler, P. H. (1998). Characterization and global distribution of vernal pools. In C. W. Witham (Ed.), *Ecology, Conservation, and Management of Vernal Pool Ecosystems* (pp. 1–14). Sacramento, USA: California Native Plant Society.

Keenan, R., Reams, G., Achard, F., De-Freitas, J., Grainger, A., & Lindquist, E. (2015). Dynamics of global forest area: results from the FAO Global Forest Resources Assessment 2015. *Forest Ecology and Management*, *352*, 9–20. http://doi.org/10.1016/j.foreco.2015.06.014

Keesing, F., Belden, L. K., Daszak, P., Dobson, A., Harvell, C. D., Holt, R. D., Hudson, P., Jolles, A., Jones, K. E., Mitchell, C. E., Myers, S. S., Bogich, T., & Ostfeld, R. S. (2010). Impacts of biodiversity on the emergence and transmission of infectious diseases. *Nature*, *468*(7324), 647–652. https://doi.org/10.1038/nature09575

Kéfi, S., Rietkerk, M., Alados, C. L., Pueyo, Y., Papanastasis, V. P., ElAich, A., & De Ruiter, P. C. (2007). Spatial vegetation patterns and imminent desertification in Mediterranean arid ecosystems. *Nature*, *449*(7159), 213–217. http://doi.org/10.1038/nature06111

Kelehear, C., Saltonstall, K., & Torchin, M. E. (2015). An introduced pentastomid parasite (*Raillietiella frenata*) infects native cane toads (*Rhinella marina*) in Panama. *Parasitology*, *142*(5), 675–679. http://doi.org/10.1017/S0031182014001759

Kelly, B. C., & Gobas, F. A. P. C. (2001). Bioaccumulation of persistent organic pollutants in lichen-caribou-wolf food chanins of Canada's central and western Arctic. *Environmental Science & Technology*, *35*(2), 325–334. http://doi.org/10.1021/es0011

Kelly, D. W. (2007). *Vectors and Pathways for Nonindigenous Aquatic Species in the Great Lakes*. Washington, DC., USA: Prepared for Committee on the St. Lawrence Seaway: Options to Eliminate Introduction of Nonindigenous Species into the Great Lakes, Phase 2 Transportation Research Board and Division on Earth and Life Studies.

Kelly, D. W., Lamberti, G. A., & MacIsaac, H. J. (2009). The Laurentian Great Lakes as a case study of biological invasion. In R. P. Keller, D. M. Lodge, M. A. Lewis, & J.F. Shogren (Eds.), *Bioeconomics of Invasive Species: Integrating Ecology, Economics, Policy and Management* (pp. 205–225). Oxford, UK: Oxford University Press.

Kembel, S. W., Cowan, P. D., Helmus, M. R., Cornwell, W. K., Morlon, H., Ackerly, D. D., Blomberg, S. P., & Webb, C. O. (2010). Picante: R tools for integrating phylogenies and ecology. *Bioinformatics*, *26*(11), 1463–1464.

Kerswell, A. P. (2006). Global biodiversity patterns of benthic marine algae. *Ecology*, *87*(10), 2479–2488. http://doi.org/10.1890/0012-9658(2006)87%5B2479:GBPOBM%5D2.0.CO;2

Kessler, M. (2001). Patterns of diversity and range size of selected plant groups along an elevational transect in the Bolivian Andes. *Biodiversity and Conservation*, *10*(11), 1897–1921. http://doi.org/10.1023/A:1013130902993

Keuroghlian, A., Andrade Santos, M. D. C., & Eaton, D. P. (2015). The effects of deforestation on white-lipped peccary (*Tayassu pecari*) home range in the southern Pantanal. *Mammalia*, *79*(4), 491–497. http://doi.org/10.1515/mammalia-2014-0094

Khoury, C. K., Bjorkman, A. D., Dempewolf, H., Ramirez-Villegas, J., Guarino, L., Jarvis, A., Rieseberg, L. H., & Struik, P. C. (2014). Increasing homogeneity in global food supplies and the implications for food security. *PNAS*, *111*(11), 4001–4006. https://doi.org/10.1073/pnas.1313490111

Kier, G., Kreft, H., Lee, T. M., Jetz, W., Ibisch, P. L., Nowicki, C., Mutke, J., & Barthlott, W. (2009). A global assessment of endemism and species richness across island and mainland regions. *PNAS*, *106*(23), 9322–9327.

Klauke, N., Schaefer, H. M., Bauer, M., & Segelbacher, G. (2016). Limited dispersal and significant fine- scale genetic structure in a tropical montane parrot species. *PLoS ONE*, *11*(12), e0169165.

Kleiber, P., Clarke, S., Bigelow, K., Nakano, H., Mcallister, M., & Takeuchi, Y. (2009). *North Pacific Blue Shark Stock Assessment*. *U.S. Department of Commerce, NOAA Technical Memorandum NMFS-PIFSC-17*. Honolulu, USA. Retrieved from https://www.pifsc.noaa.gov/tech/NOAA_Tech_Memo_PIFSC_17.pdf

Kloppenburg, J. R. (1991). *First the Seed. The political economy of plant biotechnology 1492-2000*. New York, USA: Cambridge University Press.

Knapp, A. K., Blair, J. M., Briggs, J. M., Collins, S. L., Hartnett, D. C., Johnson, L. C., & Towne, E. G. (1999). The keystone role of bison in North American tallgrass prairie: bison increase habitat heterogeneity and alter a broad array of plant, community, and ecosystem processes. *Bioscience*, *49*(1), 39–50.

Knapp, S., Dinsmore, L., Fissore, C., Hobbie, S. E., Jakobsdottir, I., Kattge, J., King, J. Y., Klotz, S., McFadden, J. P., & Cavender-Bares, J. (2012). Phylogenetic and functional characteristics of household yard floras and their changes along an urbanization gradient. *Ecology*, *93*(sp8), S83–S98. https://doi.org/10.1890/11-0392.1

Knops, J. M. H., Tilman, D., Haddad, N. M., Naeem, S., Mitchell, C. E., Haarstad, J., Ritchie, M. E., Howe, K. M., Reich, P. B., Siemann, E., & Groth, J. (1999). Effects of plant species richness on invasion dynamics, disease outbreaks, insect abundances and diversity. *Ecology Letters*, *2*(5), 286–293.

Koh, I., Lonsdorf, E. V., Williams, N. M., Brittain, C., Isaacs, R., Gibbs, J., & Ricketts, T. H. (2016). Modeling the status, trends, and impacts of wild bee abundance in the United States. *PNAS*, *113*(1), 140–145. http://doi.org/10.1073/pnas.1517685113

Köhl, M., Lasco, R., Cifuentes, M., Jonsson, Ö., Korhonen, K. T., Mundhenk, P., de Jesus Navar, J., & Stinson, G. (2015). Changes in forest production, biomass and carbon: results from the 2015 UN FAO Global Forest Resource Assessment. *Forest Ecology and Management*, *352*, 21–34. https://doi.org/10.1016/j.foreco.2015.05.036

Köhler, G. (2008). *Reptiles of Central America* (2nd ed.). Offenbach, Germany: Herpeton-Verlag.

Köhler, G. (2011). *Amphibians of Central America*. Offenbach, Germany: Herpeton-Verlag.

Kohon, J. (2011). *La Infraestructura en el Desarrollo Integral de América Latina. Diagnóstico Estratégico y Propuestas para una Agenda Prioritaria. Transporte IDeAL 2011*. Bogotá, Colombia: Corporación Andina de Fomento (CAF).

Kort, J., Richardson, J., Soolanayakanahally, R., & Schroede, W. (2014). Innovations in temperate agroforestry: the 13th North American Agroforestry Conference. *Agroforestry Systems*, *88*(4), 563–567.

Kothari, A. (2009). Protected areas and people: the future of the past. *Parks*, *17*(2), 23–34.

Kowarik, I. (2008). On the role of alien species in urban flora and vegetation. In J. M. Marzluff, E. Shulenberger, W. Endlicher, M. Alberti, G. Bradley, C. Ryan, … U. Simon (Eds.), *Urban Ecology* (pp. 321–338). Boston, USA: Springer. http://doi.org/10.1007/978-0-387-73412-5_20

Krachler, M., Zheng, J., Koerner, R., Zdanowicz, C., Fisher, D., & Shotyk, W. (2005). Increasing atmospheric antimony contamination in the northern hemisphere: snow and ice evidence from Devon Island, Arctic Canada. *Journal of Environmental Management*, *7*(12), 1169–1176. http://doi.org/10.1039/b509373b

Kraft, N. J. B., Baldwin, B. G., & Ackerly, D. D. (2010). Range size, taxon age and hotspots of neoendemism in the California flora. *Diversity and Distributions*, *16*(3), 403–413. http://doi.org/10.1111/j.1472-4642.2010.00640.x

Kramer, T., Kessler, M., Gradstein, S. R., & Acebey, A. (2005). Diversity patterns of vascular epiphytes along an elevational gradient in the Andes. *Journal of Biogeography*, *32*(10), 1799–1809. http://doi.org/10.1111/j.1365-2699.2005.01318.x

Kremen, C., & Miles, A. (2012). Ecosystem services in biologically diversified versus conventional farming systems: benefits, externalities, and trade-offs. *Ecology and Society*, *17*(4), 40. http://doi.org/10.5751/ES-05035-170440

Kroll, O., Hershler, R., Albrecht, C., Terrazas, E. M., Apaza, R., Fuentealba, C., Wolff, C., & Wilke, T. (2012). The endemic gastropod fauna of Lake Titicaca: correlation between molecular evolution and hydrographic history. *Ecology and Evolution*, *2*(7), 1517–1530. https://doi.org/10.1002/ece3.280

Krömer, T., Kessler, M., & Herzog, S. K. (2006). Distribution and flowering ecology of bromeliads along two climatically contrasting elevational transects in the Bolivian Andes. *Biotropica*, *38*(2), 183–195. http://doi.org/10.1111/j.1744-7429.2006.00124.x

Kuhlmann, M., & Ribeiro, J. F. (2016). Fruits and frugivores of the Brazilian Cerrado: ecological and phylogenetic considerations. *Acta Botanica Brasilica*, *30*(3), 495–507. http://doi.org/10.1590/0102-33062016abb0192

Küper, W., Kreft, H., Nieder, J., Köster, N., & Barthlott, W. (2004). Large-scale diversity patterns of vascular epiphytes in Neotropical montane rain forests. *Journal of Biogeography*, *31*(9), 1477–1487. http://doi.org/10.1111/j.1365-2699.2004.01093.x

Küppers, G. C., González Garraza, G. C., Quiroga, M. V., Lombardo, R., Marinone, M. C., Vinocur, A., & Mataloni, G. (2016). Drivers of highly diverse planktonic ciliate assemblages in peat bog pools from Tierra del Fuego (Argentina). *Hydrobiologia*, *773*(1), 117–134. http://doi.org/10.1007/s10750-016-2686-x

Kurz, W. A., Shaw, C. H., Boisvenue, C., Stinson, G., Metsaranta, J., Leckie, D., Dyk, A., Smyth, C., & Neilson, E. T. (2013). Carbon in Canada's boreal forest— a synthesis. *Environmental Reviews*, *21*(4), 260–292.

Kwak, M., Kami, J. A., & Gepts, P. (2009). The putative Mesoamerican domestication center of *Phaseolus vulgaris* is located in the Lerma-Santiago basin of Mexico. *Crop Science*, *49*(2), 554–563. http://doi.org/10.2135/cropsci2008.07.0421

Kyne, P. M., Carlson, J. K., Ebert, D. A., Fordham, S. V., Bizzarro, J. J., Graham, R. T., Kulka, D. W., Tewes, E. E., Harrison, L. R., & Dulvy, N. K. (Eds.). (2012). *The Conservation Status of North American, Central American, and Caribbean Chondrichthyans*. Vancouver, Canada: IUCN Species Survival Commission Shark Specialist Group.

La Sorte, F. A., & McKinney, M. L. (2007). Compositional changes over space and time along an occurrence-abundance continuum: anthropogenic homogenization of the North American avifauna. *Journal of Biogeography*, *34*(12), 2159–2167. http://doi.org/10.1111/j.1365-2699.2007.01761.x

Lähteenoja, O., Ruokolainen, K., Schulman, L., & Oinonen, M. (2009). Amazonian peatlands: an ignored C sink and potential source. *Global Change Biology*, *15*(9), 2311–2320. http://doi.org/10.1111/j.1365-2486.2009.01920.x

Lamanna, C., Blonder, B., Violle, C., Kraft, N. J. B., Sandel, B., Imova, I., Donoghue, J. C., Svenning, J.-C., McGill, B. J., Boyle, B., Buzzard, V., Dolins, S., Jorgensen, P. M., Marcuse-Kubitza, A., Morueta-Holme, N., Peet, R. K., Piel, W. H., Regetz, J., Schildhauer, M., Spencer, N., Thiers, B., Wiser, S. K., & Enquist, B. J. (2014). Functional trait space and the latitudinal diversity gradient. *PNAS*, *111*(38), 13745–13750. https://doi.org/10.1073/pnas.1317722111

Lamoreux, J. F., McKnight, M. W., & Hernandez, R. C. (2015). *Amphibian Alliance for Zero Extinction Sites in Chiapas and Oaxaca*. Gland, Switzerland: IUCN.

Lanfranco, D., & Dungey, H. S. (2001). Insect damage in *Eucalyptus*: a review of plantations in Chile. *Austral Ecology*, *26*(5), 477–481. http://doi.org/10.1046/j.1442-9993.2001.01131.x

Lannes, L. S., Bustamante, M. M. C., Edwards, P. J., & Olde Venterink, H. (2016). Native and alien herbaceous plants in the Brazilian Cerrado are (co-) limited by different nutrients. *Plant and Soil*, *400*(1–2), 231–243. http://doi.org/10.1007/s11104-015-2725-9

Lara, A., Little, C., Urrutia, R., McPhee, J., Álvarez-Garretón, C., Oyarzún, C., Soto, D., Donoso, P., Nahuelhual, L., Pino, M., & Arismendi, I. (2009). Assessment of ecosystem services as an opportunity for the conservation and management of native forests in Chile. *Forest Ecology and Management*, *258*(4), 415–424. https://doi.org/10.1016/j.foreco.2009.01.004

Larios, C., Casas, A., Vallejo, M., Moreno-Calles, A. I., & Blancas, J. (2013). Plant management and biodiversity conservation in Náhuatl homegardens of the Tehuacán Valley, Mexico. *Journal of Ethnobiology and Ethnomedicine*, *9*(74), 16. http://doi.org/10.1186/1746-4269-9-74

Larridon, I., Shaw, K., Cisternas, M. A., Paizanni Guillén, A., Sharrock, S., Oldfield, S., Goetghebeur, P., & Samain, M. S. (2014). Is there a future for the Cactaceae genera *Copiapoa*, *Eriosyce* and *Eulychnia*? A status report of a prickly situation. *Biodiversity and Conservation*, *23*(5), 1249–1287. https://doi.org/10.1007/s10531-014-0664-z

Larridon, I., Walter, H. E., Guerrero, P. C., Duarte, M., Cisternas, M. A., Hernández, C. P., Bauters, K., Asselman, P., Goetghebeur, P., & Samain, M. S. (2015). An integrative approach to understanding the evolution and diversity of *Copiapoa* (Cactaceae), a threatened endemic Chilean genus from the Atacama desert. *American Journal of Botany*, *102*(9), 1506–1520. https://doi.org/10.3732/ajb.1500168

Laurance, W. F. (2010). Habitat destruction: Death by a thousand cuts. In N. S. Sodhi & P. R. Ehrlich (Eds.), *Conservation Biology for All* (pp. 73–87). New York, USA: Oxford University Press.

Laurance, W. F., Camargo, J. L. C., Fearnside, P. M., Lovejoy, T. E., Williamson, G. B., Mesquita, R. C. G., Meyer, C. F. J., Bobrowiec, P. E. D., & Laurance, S. G. W. (2017). An Amazonian rainforest and its fragments as a laboratory of global change. *Biological Review*. https://doi.org/10.1111/brv.12343

Laurance, W. F., Goosem, M., & Laurance, S. G. W. (2009). Impacts of roads and linear clearings on tropical forests. *Trends in Ecology and Evolution*, *24*(12), 659–669. http://doi.org/10.1016/j.tree.2009.06.009

Lawrence, D., & Vandecar, K. (2015). Effects of tropical deforestation on climate and agriculture. *Nature*

Climate Change, *5*(1), 27–36. http://doi.org/10.1038/nclimate2430

Leach, M. K., & Givnish, T. J. (1996). Ecological determinants of species loss in remnant prairies. *Science*, *273*(5281), 1555–1558. http://doi.org/10.1126/science.273.5281.1555

Leal-Flórez, J. (2008). *Impacts of non-native fishes on the fish community and the fishery of the Ciénaga Grande de Santa Marta estuary, northern Colombia*. Doctoral dissertation, Ph. D. Thesis, University of Bremen.

Leão, Z., R. Kikuchi, and M. Oliveira. 2010. Status of Eastern Brazilian coral reefs in time of climate changes. *Pan-American Journal of Aquatic Sciences,* 5:224-235.

Le Maitre, D. C., Gush, M. B., & Dzikiti, S. (2015). Impacts of invading alien plant species on water flows at stand and catchment scales. *AoB Plants*, *7*(1), plv043. http://doi.org/10.1093/aobpla/plv043

Le Saout, S., Hoffmann, M., Shi, Y., Hughes, A., Bernard, C., Brooks, T. M., Bertzky, B., Butchart, S. H. M., Stuart, S. N., Badman, T., & Rodrigues, A. S. L. (2013). Protected areas and effective biodiversity conservation. *Science*, *342*(6160), 803–805. https://doi.org/10.1126/science.1239268

Lefcheck, J. S., Byrnes, J. E. K., Isbell, F., Gamfeldt, L., Griffin, J. N., Eisenhauer, N., Hensel, M. J. S., Hector, A., Cardinale, B. J., & Duffy, J. E. (2015). Biodiversity enhances ecosystem multifunctionality across trophic levels and habitats. *Nature Communications*, *6*, 6936. https://doi.org/10.1038/ncomms7936

Leishman, M. R., Westoby, M., & Jurado, E. (1995). Correlates of seed size variation: a comparison among five temperate floras. *Journal of Ecology*, *83*(3), 517–529. http://doi.org/10.2307/2261604

León, O. A., & Vargas-Ríos, O. (2011). Estrategias para el control, manejo y restauración de áreas invadidas por retamo espinoso (Ulex europaeus) en la vereda El Hato, localidad de Usme, Bogotá D.C. In O. Vargas & S. Reyes (Eds.), *La Restauración Ecológica en la Practica: Memorias del I Congreso Colombiano de Restauración Ecológica* (pp. 474–490). Bogotá, D.C., Colombia: Universidad Nacional de Colombia.

Leslie, H. M. (2008). Global coastal change. *Quarterly Review of Biology*, *83*(1), 136–137. http://doi.org/10.4031/002533206787353278

Lester, E. D., Satomi, M., & Ponce, A. (2007). Microflora of extreme arid Atacama Desert soils. *Soil Biology and Biochemistry*, *39*(2), 704–708. http://doi.org/10.1016/j.soilbio.2006.09.020

Letourneau, D. K., Armbrecht, I., Rivera, B. S., Lerma, J., Carmona, E. J., Daza, M. C., Escobar, S., Galindo, V., GutiéRrez, C., LóPez, S. D., Mejía, J. L., Rangel, A. M. A., Rangel, J. H., Rivera, L., Saavedra, C. A., Torres, A. M., & Trujillo, A. R. (2011). Does plant diversity benefit agroecosystems? A synthetic review. *Ecological Applications*. https://doi.org/10.1890/09-2026.1

Leveau, L. M. (2013). Bird traits in urban– rural gradients: how many functional groups are there? *Journal of Ornithology*, *154*(3), 655–662. http://doi.org/10.1007/s10336-012-0928-x

Liang, J., Crowther, T. W., Picard, N., Wiser, S., Zhou, M., Alberti, G., Schulze, E.-D., McGuire, A. D., Bozzato, F., Pretzsch, H., De-Miguel, S., Paquette, A., Herault, B., Scherer-Lorenzen, M., Barrett, C. B., Glick, H. B., Hengeveld, G. M., Nabuurs, G.-J., Pfautsch, S., Viana, H., Vibrans, A. C., Ammer, C., Schall, P., Verbyla, D., Tchebakova, N., Fischer, M., Watson, J. V., Chen, H. Y. H., Lei, X., Schelhaas, M.-J., Lu, H., Gianelle, D., Parfenova, E. I., Salas, C., Lee, E., Lee, B., Kim, H. S., Bruelheide, H., Coomes, D. A., Piotto, D., Sunderland, T., Schmid, B., Gourlet-Fleury, S., Sonke, B., Tavani, R., Zhu, J., Brandl, S., Vayreda, J., Kitahara, F., Searle, E. B., Neldner, V. J., Ngugi, M. R., Baraloto, C., Frizzera, L., Ba azy, R., Oleksyn, J., Zawi a-Nied wiecki, T., Bouriaud, O., Bussotti, F., Finer, L., Jaroszewicz, B., Jucker, T., Valladares, F., Jagodzinski, A. M., Peri, P. L., Gonmadje, C., Marthy, W., OBrien, T., Martin, E. H., Marshall, A. R., Rovero, F., Bitariho, R., Niklaus, P. A., Alvarez-Loayza, P., Chamuya, N., Valencia, R., Mortier, F., Wortel, V., Engone-Obiang, N. L., Ferreira, L. V., Odeke, D. E., Vasquez, R. M., Lewis, S. L., & Reich, P. B. (2016). Positive biodiversity-productivity relationship predominant in global forests. *Science*, *354*(6309), aaf8957. https://doi.org/10.1126/science.aaf8957

Liljedahl, A. K., Boike, J., Daanen, R. P., Fedorov, A. N., Frost, G. V., Grosse, G., Hinzman, L. D., Iijma, Y., Jorgenson, J. C., Matveyeva, N., Necsoiu, M., Raynolds, M. K., Romanovsky, V. E., Schulla, J., Tape, K. D., Walker, D. A., Wilson, C. J., Yabuki, H., & Zona, D. (2016). Pan-Arctic ice-wedge degradation in warming permafrost and its influence on tundra hydrology. *Nature Geoscience*, *9*(4), 312–318. https://doi.org/10.1038/ngeo2674

Linares-Palomino, R., Oliveira-Filho, A. T., & Pennington, R. T. (2011). Neotropical seasonally dry forests: diversity, endemism and biogeography of woody plants. In R. Dirzo, H. Young, H. Mooney, & G. Ceballos (Eds.), *Seasonally Dry Tropical Forests: Ecology and Conservation* (pp. 3–21). Washington, DC, USA: Island Press.

Lind, E. M., Vincent, J. B., Weiblen, G. D., Cavender-Bares, J. M., & Borer, E. T. (2015). Trophic phylogenetics: evolutionary influences on body size, feeding, and species associations in grassland arthropods. *Ecology*, *96*(4), 998–1009.

Lira, P. K., Tambosi, L. R., Ewers, R. M., & Metzger, J. P. (2012). Land-use and land-cover change in Atlantic Forest landscapes. *Forest Ecology and Management*, *278*, 80–89. http://doi.org/10.1016/j.foreco.2012.05.008

Liu, Y., Pan, X., & Li, J. (2015). A 1961-2010 record of fertilizer use, pesticide application and cereal yields: a review. *Agronomy for Sustainable Development*, *35*(1), 83–93. http://doi.org/10.1007/s13593-014-0259-9

Lloyd, A. H., & Bunn, A. G. (2007). Responses of the circumpolar boreal forest to 20th century climate variability. *Environmental Research Letters*, *2*(4), 45013. Retrieved from http://stacks.iop.org/1748-9326/2/i=4/a=045013

Loehle, C. (2000). Strategy space and the disturbance spectrum: a life-history model for tree species coexistence. *The American Naturalist*, *156*(1), 14–33. http://doi.org/10.1086/303369

Loh, E. H., Zambrana-Torrelio, C., Olival, K. J., Bogich, T. L., Johnson, C. K., Mazet, J. A. K., Karesh, W., & Daszak, P. (2015). Targeting transmission pathways for emerging zoonotic disease surveillance and control. *Vector-Borne and Zoonotic Diseases*, *15*(7), 432–437. https://doi.org/10.1089/vbz.2013.1563

Loisel, J., & Yu, Z. (2013). Holocene peatland carbon dynamics in Patagonia. *Quaternary Science Reviews*, *69*, 125–141. http://doi.org/10.1016/j.quascirev.2013.02.023

Londoño, C., Cleef, A., & Madriñán, S. (2014). Angiosperm flora and biogeography of the páramo region of Colombia, Northern Andes. *Flora: Morphology, Distribution, Functional Ecology of Plants*, *209*(2), 81–87. http://doi.org/10.1016/j.flora.2013.11.006

López-Austin, A. (1990). *Cuerpo Humano e Ideología. Las Concepciones de los Antiguos Nahuas (2 vols)*. Mexico City, Mexico: Universidad Nacional Autónoma de México, Instituto de Investigaciones Antropológicas.

Loreau, M. (2004). Does functional redundancy exist? *Oikos*, *104*(3), 606–611. http://doi.org/10.1111/j.0030-1299.2004.12685.x

Loreau, M. (2010). *From Populations to Ecosystems: Theoretical Foundations for a New Ecological Synthesis*. New Jersey, USA: Princeton University Press.

Loreau, M., & Hector, A. (2001). Partitioning selection and complementarity in biodiversity experiments. *Nature*, *412*(6842), 72–76. http://doi.org/10.1038/35083573

Lucifora, L. O., García, V. B., & Worm, B. (2011). Global diversity hotspots and conservation priorities for sharks. *PLoS ONE*, *6*(5), e19356. http://doi.org/10.1371/journal.pone.0019356

Luebert, F., & Pliscoff, P. (2006). *Sinopsis Bioclimática y Vegetacional de Chile*. Santiago, Chile: Editorial Universitaria.

Lughadha, E. N., Govaerts, R., Belyaeva, I., Black, N., Lindon, H., Allkin, R., Magill, R. E., & Nicolson, N. (2016). Counting counts: revised estimates of numbers of accepted species of flowering plants, seed plants, vascular plants and land plants with a review of other recent estimates. *Phytotaxa*, *272*(1), 82–88. https://doi.org/10.11646/phytotaxa.272.1.5

Lugo, A. E., Helmer, E. H., & Valentín, E. S. (2012). Caribbean landscapes and their biodiversity. *Interciencia*, *37*(9), 705–710.

Luteyn, J. L. (1999). *Páramos: A Checklist of Plant Diversity, Geographical Distribution, and Botanical Literature*. New York, USA: The New York Botanical Garden Press.

Lutz, D. A., Powell, R. L., & Silman, M. R. (2013). Four decades of Andean timberline migration and implications for biodiversity loss with climate change. *PLoS ONE*, *8*(9), e74496. http://doi.org/10.1371/journal.pone.0074496

Lyra-Jorge, M. C., Ciocheti, G., & Pivello, V. R. (2008). Carnivore mammals in a fragmented landscape in northeast of Sao Paulo State, Brazil. *Biodiversity and Conservation*, *17*(7), 1573–1580.

MacGregor-Fors, I., & Ortega-Álvarez, R. (Eds.). (2013). Ecología Urbana: Experiencias en América Latina. Retrieved from www1.inecol.edu.mx/libro_ecologia_urbana

Macossay-Cortez, A., Sánchez, A. J., Florido, R., Huidobro, L., & Montalvo-Urgel, H. (2011). Historical and environmental distribution of ichthyofauna in the tropical wetland of Pantanos de Centla, southern Gulf of Mexico. *Acta Ichthyologica et Piscatoria*, *41*(3), 229–245. http://doi.org/10.3750/AIP2011.41.3.11

Maestre, F. T., Quero, J. L., Gotelli, N. J., Escudero, A., Ochoa, V., Delgado-Baquerizo, M., García-Gómez, M., Bowker, M. A., Soliveres, S., Escolar, C., García-Palacios, P., Berdugo, M., Valencia, E., Gozalo, B., Gallardo, A., Aguilera, L., Arredondo, T., Blones, J., Boeken, B., Bran, D., Conceição, A. A., Cabrera, O., Chaieb, M., Derak, M., Eldridge, D. J., Espinosa, C. I., Florentino, A., Gaitán, J., Gatica, M. G., Ghiloufi, W., Gómez-González, S., Gutiérrez, J. R., Hernández, R. M., Huang, X., Huber-Sannwald, E., Jankju, M., Miriti, M., Monerris, J., Mau, R. L., Morici, E., Naseri, K., Ospina, A., Polo, V., Prina, A., Pucheta, E., Ramírez-Collantes, D. A., Romão, R., Tighe, M., Torres-Díaz, C., Val, J., Veiga, J. P., Wang, D., & Zaady, E. (2012). Plant species richness and ecosystem multifunctionality in global drylands. *Science*, *335*(6065), 214–218. https://doi.org/10.1126/science.1215442

Maitner, B. S., Boyle, B., Casler, N., Condit, R., Donoghue, J., Durán, S. M., Guaderrama, D., Hinchliff, C. E., Jørgensen, P. M., Kraft, N. J. B., McGill, B., Merow, C., Morueta-Holme, N., Peet, R. K., Sandel, B., Schildhauer, M., Smith, S. A., Svenning, J.-C., Thiers, B., Violle, C., Wiser, S., & Enquist, B. J. (2017). The BIEB r package: A tool to access the Botanical Information and Ecology Network (BIEN) database. *Methods in Ecology and Evolution*, (March), 1–7. https://doi.org/10.1111/2041-210X.12861

Malanson, G. P., Cheney, A. B., & Kinney, M. (2015). Climatic and geographic relations of alpine tundra floras in western North America. *Alpine Botany*, *125*(1), 21–29. http://doi.org/10.1007/s00035-014-0144-9

Malanson, G. P., Zimmerman, D. L., & Fagre, D. B. (2015). Floristic similarity, diversity and endemism as indicators of refugia characteristics and needs in the West. *Biodiversity*, *16*(4), 237–246. http://doi.org/10.1080/14888386.2015.1117989

Manhães, M. A., & Loures-Ribeiro, A. (2005). Spatial distribution and diversity of bird community in an urban area of southeast Brazil. *Brazilian Archives of Biology and Technology*, *48*(2), 285–294. http://doi.org/10.1590/S1516-89132005000200016

Maragliano, R. E., Marti, L. J., Ibañez, L. M., & Montalti, D. (2009). Comunidades de aves urbanas de Lavallol, Buenos Aires, Argentina. *Acta Zoologica Lilloana*, *53*(1–2), 108–114.

Marateo, G., Grilli, P., Bouzas, N., Jensen, R., Ferretti, V., Juarez, M., & Soave, G. (2013). Uso de hábitat por aves en rellenos sanitarios del noreste de la provincia de Buenos Aires, Argentina. *Ecologia Austral*, (Diciembre 2013), 202–208.

Mares, M. (1992). Neotropical mammals and the myth of Amazonian biodiversity. *Science*, *255*(5047), 976–979. http://doi.org/10.1126/science.255.5047.976

Marini, M. Â., & García, F. I. (2005). Bird conservation in Brazil. *Conservation Biology*, *19*(3), 665–671. http://doi.org/10.1111/j.1523-1739.2005.00706.x

Marini, M. Â. (2001). Effects of forest fragmentation on birds of the cerrado region, Brazil. *Bird Conservation International*, *11*(1), 13–25. http://doi.org/10.1017/S0959270901001034

Marquard, E., Weigelt, A., Temperton, V. M., Roscher, C., Schumacher, J., Buchmann, N., Fischer, M., Weisser, W. W., & Schmid, B. (2009). Plant species richness and functional composition drive overyielding in a six-year grassland experiment. *Ecology*, *90*(12), 3290–3302. https://doi.org/10.1890/09-0069.1

Martín-Forés, I., Sánchez-Jardón, L., Acosta-Gallo, B., del Pozo, A., Castro, I., de Miguel, J. M., Ovalle, C., & Casado, M. A. (2015). From Spain to Chile: environmental filters and success of herbaceous species in Mediterranean-climate regions. *Biological Invasions*, *17*(5), 1425–1438. https://doi.org/10.1007/s10530-014-0805-z

Martin, L. M., Moloney, K. A., & Wilsey, B. J. (2005). An assessment of grassland restoration success using species diversity components. *Journal of Applied Ecology*, *42*(2), 327–336. http://doi.org/10.1111/j.1365-2664.2005.01019.x

Martin, S. (2008). Global diversity of crocodiles (Crocodilia, Reptilia) in freshwater. *Hydrobiologia*, *595*(1), 587–591. http://doi.org/10.1007/s10750-007-9030-4

Martyniuk, N. A., Morales, C. L., & Aizen, M. A. (2015). Invasive conifers reduce seed set of a native Andean cedar through heterospecific pollination competition. *Biological Invasions*, *17*(4), 1055–1067. http://doi.org/10.1007/s10530-014-0775-1

Mascitti, V., & Bonaventura, S. M. (2002). Patterns of abundance, distribution and habitat use of flamingos in the high Andes, South America. *Waterbirds*, *25*, 358–365. http://doi.org/10.1675/1524-4695(2002)025%5B0358:POADAH%5D2.0.CO;2

Masera, O. R., Bellon, M. R., & Segura, G. (1995). Forest management options for sequestering carbon in Mexico. *Biomass and Bioenergy*, *8*(5), 357–367. http://doi.org/10.1016/0961-9534(95)00028-3

Matheson, K., McKenzie, C. H., Gregory, R. S., Robichaud, D. A., Bradbury, I. R., Snelgrove, P. V. R., & Rose, G. A. (2016). Linking eelgrass decline and impacts on associated fish communities to European green crab *Carcinus maenas* invasion. *Marine Ecology Progress Series*, *548*, 31–45. http://doi.org/10.3354/meps11674

Maxwell, S. L., Fuller, R. A., Brooks, T. M., & Watson, J. E. M. (2016). Biodiversity: The ravages of guns, nets and bulldozers. *Nature*, *536*(7615), 143–145. http://doi.org/10.1038/536143a

McCarthy, A. J., Shaw, M.-A., & Goodman, S. J. (2007). Pathogen evolution and disease emergence in carnivores. *Proceedings of the Royal Society B: Biological Sciences*, *274*(1629), 3165–3174. http://doi.org/10.1098/rspb.2007.0884

McCauley, D. J., Pinsky, M. L., Palumbi, S. R., Estes, J. A., Joyce, F. H., & Warner, R. R. (2015). Marine defaunation: Animal loss in the global ocean. *Science*, *347*(6219), 1255641 (1-7). http://doi.org/10.1126/science.1255641

McCleery, R. (2010). Urban mammals. In J. Aitkenhead-Peterson & A. Volder (Eds.), *Urban Ecosystem Ecology, Agronomy Monographs 55* (pp. 87–102). Madison, USA: American Society of Agronomy, Crop Science Society of America, Soil Science Society of America.

McIntyre, N. E. (2000). Ecology of urban arthropods: A review and a call to action. *Annals of the Entomological Society of America*, *93*(4), 825–835. http://doi.org/10.1603/0013-8746(2000)093%5B0825:EOUAAR%5D2.0.CO;2

McIntyre, N. E., Rango, J., Fagan, W. F., & Faeth, S. H. (2001). Ground arthropod community structure in a heterogeneous urban environment. *Landscape and Urban Planning*, *52*(4), 257–274. http://doi.org/10.1016/S0169-2046(00)00122-5

McIntyre, P. B., Reidy Liermann, C. A., & Revenga, C. (2016). Linking freshwater fishery management to global food security and biodiversity conservation. *PNAS*, *113*(45), 12880–12885. http://doi.org/10.1073/pnas.1521540113

McIntyre, P. J., Thorne, J. H., Dolanc, C. R., Flint, A. L., Flint, L. E., Kelly, M., & Ackerly, D. D. (2015). Twentieth-century shifts in forest structure in California: Denser forests, smaller trees, and increased dominance of oaks. *PNAS*, *112*(5), 1458–1463. http://doi.org/10.1073/pnas.1410186112

McKane, R. B., Johnson, L. C., Shaver, G. R., Nadelhoffer, K. J., Rastetter, E. B., Fry, B., Giblin, A. E., Kielland, K., Kwiatkowski, B. L., Laundre, J. A., & Murray, G. (2002). Resource-based niches provide a basis for plant species diversity and dominance in arctic tundra. *Nature*, *415*(6867), 68–71. https://doi.org/10.1038/415068a

McKee, K. L., Cahoon, D. R., & Feller, I. C. (2007). Caribbean mangroves adjust to rising sea level through biotic controls on change in soil elevation. *Global Ecology and Biogeography*, *16*, 545–556. http://doi.org/10.1111/j.1466-8238.2007.00317.x

McKinney, M. L. (2002). Urbanization, biodiversity, and conservation. *BioScience*, *52*(10), 883–890. http://doi.org/10.1641/0006-3568(2002)052%5B0883:UBAC%5D2.0.CO;2

McKinney, M. L. (2006). Urbanization as a major cause of biotic homogenization. *Biological Conservation*, *127*, 247–260. http://doi.org/10.1016/j.biocon.2005.09.005

McKinney, M. L. (2008). Effects of urbanization on species richness: A review of plants and animals. *Urban Ecosystems*, *11*(2), 161–176. http://doi.org/10.1007/s11252-007-0045-4

McNaughton, S. J., Oesterheld, M., Frank, D. A., & Williams, K. J. (1989). Ecosystem-level patterns of primary productivity and herbivory in terrestrial habitats. *Nature*, *341*, 142–144. http://doi.org/10.1038/341142a0

McPhearson, T., Auch, R., & Alberti, M. (2013). Regional assessment of North America: Urbanization trends, biodiversity patterns, and ecosystem services. In T. Elmqvist, M. Fragkias, J. Goodness, B. Güneralp, P. J. Marcotullio, R. I. McDonald, … C. Wilkinson (Eds.), *Urbanization, Biodiversity and Ecosystem Services: Challenges and Opportunities: A Global Assessment* (pp. 279–286). New York, USA: Springer.

MD Sea Grant. (n.d.). Bait Worm Study. Retrieved from http://www.mdsg.umd.edu/topics/aquatic-invasive-species/bait-worm-study

Medan, D., Torretta, J. P., Hodara, K., de la Fuente, E. B., & Montaldo, N. H. (2011). Effects of agriculture expansion and intensification on the vertebrate and invertebrate diversity in the Pampas of Argentina. *Biodiversity and Conservation*, *20*(13), 3077–3100. http://doi.org/10.1007/s10531-011-0118-9

Medeiros, R. B. de, Focht, T., Menegon, L. L., & Freitas, M. R. (2014). Seed longevity of *Eragrostis plana* Nees buried in natural grassland soil. *Revista Brasileira de Zootecnia*, *43*(11), 561–567. http://doi.org/10.1590/S1516-35982014001100001

Meltofte, H. (Ed.). (2013). *Arctic Biodiversity Assessment. Status and Trends in Arctic Biodiversity*. Akureyri, Iceland: Conservation of Arctic Flora and Fauna.

Mena-Torres, F., Fernández-San Juan, M., Campos, B., Sánchez-Avila, J., Faría, M., Pinnock-Branford, M. V., de la Cruz-Malavassi, E. M., Lacorte, S., Soares, A. M. V. M., & Barata., C. (2014). Pesticide residue analyses and biomarker responses of native Costa Rican fish of the Poeciliidae and Cichlidae families to assess environmental impacts of pesticides in Palo Verde National Park. *Journal of Environmental Biology*, *35*(1), 19–27.

Mena, J. L., Williams, M., Gazzolo, C., & Montero, F. (2007). Estado de conservación de *Melanomys zunigae* (Sanborn 1949) y de los mamíferos pequeños en las Lomas de Lima. *Revista Peruana de Biología*, *14*(2), 201–207.

Mendenhall, C. D., Karp, D. S., Meyer, C. F. J., Hadly, E. A., & Daily, G. C. (2014). Predicting biodiversity change and averting collapse in agricultural landscapes. *Nature*, *509*(7499), 213–217. http://doi.org/10.1038/nature13139

Mendoza, R., & Koleff, P. (2014). *Especies acuáticas invasoras en México*. Mexico City, Mexico: Comisión Nacional para el Conocimiento y Uso de la Biodiversidad, México.

Menominee Tribal Enterprises. (2012). Forest Management Plan (Revised 1973) 2012-2027. November 29, 2012. Retrieved from http://www.mtewood.com/Forestry/FMP/finalfmp06112012.pdf

Meserve, P. L., Kelt, D. A., Previtali, M. A., Milstead, W. B., & Gutiérrez, J. R. (2011). Global climate change and small mammal populations in north-central Chile. *Journal of Mammalogy*, *92*(6), 1223–1235. http://doi.org/10.1644/10-MAMM-S-267.1

Metzger, J. P. (2009). Conservation issues in the Brazilian Atlantic forest. *Biological Conservation*, *142*(6), 1138–1140. http://doi.org/10.1016/j.biocon.2008.10.012

Michaelson, G. J., Ping, C. L., & Kimble, J. M. (1996). Carbon storage and distribution in tundra soils of Arctic Alaska, U.S.A. *Arctic and Alpine Research*, *28*(4), 414–424. http://doi.org/10.2307/1551852

Millennium Ecosystem Assessment. (2005). *Ecosystems and Human Well-being*. Washington, D.C., USA: Island Press.

Milliman, J. D., Farnsworth, K. L., Jones, P. D., Xu, K. H., & Smith, L. C. (2008). Climatic and anthropogenic factors affecting river discharge to the global ocean, 1951-2000. *Global and Planetary Change*, *62*(3), 187–194. http://doi.org/10.1016/j.gloplacha.2008.03.001

Miloslavich, P., Díaz, J. M., Klein, E., Alvarado, J. J., Díaz, C., Gobin, J., Escobar-Briones, E., Cruz-Motta, J. J., Weil, E., Cortés, J., Bastidas, A. C., Robertson, R., Zapata, F., Martín, A., Castillo, J., Kazandjian, A., & Ortiz, M. (2010). Marine biodiversity in the Caribbean: regional estimates and distribution patterns. *PLoS ONE*, *5*(8), e11916. https://doi.org/10.1371/journal.pone.0011916

Miloslavich, P., Klein, E., Diaz, J. M., Hernandez, C. E., Bigatti, G., Campos, L., Artigas, F., Castillo, J., Penchaszadeh, P. E., Neill, P. E., Carranza, A., Retana, M. V., de Astarloa, J. M. D., Lewis, M., Yorio, P., Piriz, M. L., Rodriguez, D., Yoneshigue-Valentin, Y., Gamboa, L., & Martin, A. (2011). Marine biodiversity in the Atlantic and Pacific Coasts of South America: Knowledge and gaps. *PLoS ONE*, *6*(1), e14631. https://doi.org/10.1371/journal.pone.0014631

Ministerio del Ambiente. (2014). *Plan Nacional de Restauración Forestal 2014-2017*. Quito, Ecuador. Retrieved from

http://sociobosque.ambiente.gob.ec/files/images/articulos/archivos/amrPlanRF.pdf
Ministerio del Medio Ambiente. (2017). No Title.
Ministério do Meio Ambiente (MMA). (2006). *Programa REVIZEE. Avaliação do Potencial Sustentável de Recursos Vivos na Zona Econômica Exclusiva*. Brasília, Brazil: Relatório Executivo.
Mites, M. (2008). Criteria used to set export quotas for Appendix I and II orchid species from Ecuador. NDF Workshop WG 4 – Geophytes and epiphytes. Case study 3 summary. Retrieved from https://cites.org/sites/default/files/ndf_material/WG4-CS3-S.pdf
Mittermeier, R. A., Gil, P. R., Hoffmann, M., Pilgrim, J., Brooks, T., Mittermeier, C. G., Lamoreux, J., & Fonseca, G. A. (2005). *Hotspots Revisited: Earth's Biologically Richest and Most Endangered Terrestrial Ecoregions*. Mexico City, Mexico: CEMEX Books on Nature Series.
Moffat, N. D., Lantz, T. C., Fraser, R. H., & Olthof, I. (2016). Recent vegetation change (1980–2013) in the tundra ccosystems of the Tuktoyaktuk Coastlands, NWT, Canada. *Arctic, Antarctic, and Alpine Research*, *48*(3), 581–597. http://doi.org/10.1657/AAAR0015-063
Moles, A. T., Ackerly, D. D., Webb, C. O., Tweddle, J. C., Dickie, J. B., & Westoby, M. (2005). A brief history of seed size. *Science*, *307*(5709), 576–580. http://doi.org/10.1126/science.1104863
Molnar, J. L., Gamboa, R. L., Revenga, C., & Spalding, M. D. (2008). Assessing the global threat of invasive species to marine biodiversity. *Frontiers in Ecology and the Environment*, *6*(9), 485–492. http://doi.org/10.1890/070064
Monigatti, M., Bussmann, R. W., & Weckerle, C. S. (2013). Medicinal plant use in two Andean communities located at different altitudes in the Bolivar Province, Peru. *Journal of Ethnopharmacology*, *145*(2), 450–464. http://doi.org/10.1016/j.jep.2012.10.066
Montalva, J., Castro, B., & Allendes, J. L. (2010). Las abejas (Hymenoptera: Apoidea) del Jardín Botánico Chagual. Estudio de caso de abejas nativas en zonas urbanas de Santiago de Chile. *Revista Del Jardín Botánico Chagual*, *No. 8*, 13–23.
Montalva, J., Dudley, L., Arroyo, M. K., Retamales, H., & Abrahamovich, A. H. (2011). Geographic distribution and associated flora of native and introduced bumble bees (*Bombus* spp.) in Chile. *Journal of Apicultural Research*, *50*(1), 11–21. http://doi.org/10.3896/IBRA.1.50.1.02
Montalva, J., & Ruz, L. (2010). Actualización a la lista sistemática de las abejas chilenas. *Revista Chilena de Entomología*, *8*, 13–23.
Montecino, V., Molina, X., Kumar, S., Castillo, M. L. C., & Bustamante, R. O. (2014). Niche dynamics and potential geographic distribution of *Didymosphenia geminata* (Lyngbye) M. Schmidt, an invasive freshwater diatom in Southern Chile. *Aquatic Invasions*, *9*(4), 507–519. http://doi.org/10.3391/ai.2014.9.4.09
Montenegro, G., Gómez, M., Díaz, F., & Ginocchio, R. (2003). Regeneration potencial of Chilean matorral after fire. In T. T. Veblen, W. L. Baker, G. Montenegro, & T. W. Swetnam (Eds.), *Fire and Climatic Change in Temperate Ecosystems of the Western Americas* (pp. 381–409). New York: Springer.
Montes-Leyva, L., Téllez-Valdés, O., Bojorquez, L., Dávila, P., & Lira, R. (2017). Potential areas for conservation of useful flora of the Tehuacán-Cuicatlán Valley, Mexico. *Genetic Resources and Crop Evolution*, *65*(1), 343–354. http://doi.org/10.1007/s10722-017-0538-9
Moore, A. A., & Palmer, M. A. (2005). Invertebrate biodiversity in agricultural and urban headwater streams: Implications for conservation and management. *Ecological Applications*, *15*(4), 1169–1177. http://doi.org/10.1890/04-1484
Morales, C. L., Arbetman, M. P., Cameron, S. A., & Aizen., M. A. (2013). Rapid ecological replacement of a native bumble bee by invasive species. *Frontiers in Ecology and the Environment*, *11*(10), 529–534. http://doi.org/10.1890/120321
Moreno-Calles, A. I., Casas, A., Rivero-Romero, A. D., Romero-Bautista, Y. A., Rangel-Landa, S., Fisher-Ortíz, R. A., Alvarado-Ramos, F., Vallejo-Ramos, M., & Santos-Fita, D. (2016). Ethnoagroforestry: integration of biocultural diversity for food sovereignty in Mexico. *Journal of Ethnobiology and Ethnomedicine*, *12*, 54. https://doi.org/10.1186/s13002-016-0127-6
Moreno-Calles, A. I., Casas, A., Toledo, V. M., & Vallejo-Ramos, M. (Eds.). (2016). *Etnoagroforestería en México*. México D.F., México: Universidad Nacional Autónoma de México.
Moret, P., Aráuz, M., Gobbi, M., & Barragán, Á. (2016). Climate warming effects in the tropical Andes: first evidence for upslope shifts of Carabidae (Coleoptera) in Ecuador. *Insect Conservation and*

Diversity, *9*(4), 342–350. http://doi.org/10.1111/icad.12173

Moritz, C. (2002). Strategies to protect biological diversity and the evolutionary processes that sustain it. *Systematic Biology*, *51*(2), 238–254. http://doi.org/10.1080/10635150252899752

Moritz, C., Patton, J. L., Conroy, C. J., Parra, J. L., White, G. C., & Beissinger, S. R. (2008). Impact of a century of climate change on small-mammal communities in Yosemite National Park, USA. *Science*, *322*(5899), 261–264. http://doi.org/10.1126/science.1163428

Moro, M. F., Lughadha, E., Filer, D., Soares de Araújo, F., & Martins, F. (2014). A catalogue of the vascular plants of the Caatinga Phytogeographical Domain: a synthesis of floristic and phytosociological surveys. *Phytotaxa*, *160*(1), 1–118. http://doi.org/10.11646/phytotaxa.160.1.1

Morón, A., & Sawchik, J. (2003). Soil quality indicators in a long-term crop-pasture rotation experiment in Uruguay. In *17th World Congress of Soil Science Symposium n° 32* (p. 1327). Serie Técnica 134. INIA La Estanzuela.

Morón Zambrano, V. I., García Rangel, S., & Yerena, E. (2015). *Deforestación en Venezuela: Una Comparación de las Evaluaciones Existentes*. Caracas, Venezuela. Retrieved from https://www.researchgate.net/profile/Vilisa_Moron_Zambrano/publication/301889738_Deforestacion_en_Venezuela_Una_comparacion_de_las_evaluaciones_existentes/links/572b32f608ae057b0a094add/Deforestacion-en-Venezuela-Una-comparacion-de-las-evaluaciones-exist

Morueta-Holme, N., Engemann, K., Sandoval-Acuña, P., Jonas, J. D., Segnitz, R. M., & Svenning, J.-C. (2015). Strong upslope shifts in Chimborazo's vegetation over two centuries since Humboldt. *PNAS*, *112*(41), 12741–12745. http://doi.org/10.1073/pnas.1509938112

Müller, N., Ignatieva, M., Nilon, C. H., Werner, P., & Zipperer, W. C. (2013). Patterns and trends in urban biodiversity and landscape design. In *Urbanization, Biodiversity and Ecosystem Services: Challenges and Opportunities: A Global Assessment* (pp. 123–174). http://doi.org/10.1007/978-94-007-7088-1_10

Mumby, P. J., Dahlgren, C. P., Harborne, A. R., Kappel, C. V., Micheli, F., Brumbaugh, D. R., Holmes, K. E., Mendes, J. M., Broad, K., Sanchirico, J. N., Buch, K., Box, S., Stoffle, R. W., & Gill, A. B. (2006). Fishing, trophic cascades, and the process of grazing on coral reefs. *Science*, *311*(5757), 98–101. https://doi.org/10.1126/science.1121129

Muñoz-Concha, D., Farías, C., & Méndez, J. (2015). Notes on a new population of the endangered Chilean tree *Gomortega keule*. *New Zealand Journal of Botany*, *53*(4), 224–230. http://doi.org/10.1080/0028825X.2015.1064974

Muñoz-Pedreros, A., de los Rios-Escalante, P., & Möller, P. (2015). Zooplankton of the highland bogs of Putana, a desert wetland of the high puna, northern Chile. *Crustaceana*, *88*(10–11), 1235–1244. http://doi.org/DOI 10.1163/15685403-00003482

Muñoz, A. A., & Cavieres, L. A. (2008). The presence of a showy invasive plant disrupts pollinator service and reproductive output in native alpine species only at high densities. *Journal of Ecology*, *96*(3), 459–467. http://doi.org/10.1111/j.1365-2745.2008.01361.x

Muñoz, Y., Cadena, A., & Rangel-Ch., J. O. (2000). Mamíferos. In J. O. Rangel Ch. (Ed.), *Colombia Diversidad Biótica III* (pp. 599–611). Santafé de Bogotá, D.C., Colombia: Universidad Nacional de Colombia. Retrieved from http://www.uneditorial.net/pdf/TomoIII.pdf

Myers-Smith, I. H., Elmendorf, S. C., Beck, P. S., Wilmking, M., Hallinger, M., Blok, D., Tape, K. D., Rayback, S. A., Macias-Fauria, M., Speed, J. D., & Vellend, M. (2015). Climate sensitivity of shrub growth across the tundra biome. *Nature Climate Change*, *5*(9), 887–891. https://doi.org/10.1038/NCLIMATE2697

Myers-Smith, I. H., Forbes, B. C., Wilmking, M., Hallinger, M., Lantz, T., Blok, D., Tape, K. D., Macias-Fauria, M., Sass-Klaassen, U., Lévesque, E., Boudreau, S., Ropars, P., Hermanutz, L., Trant, A., Siegwart Collier, L., Weijers, S., Rozema, J., Rayback, S. A., Martin Schmidt, N., Schaepman-Strub, G., Wipf, S., Rixen, C., Ménard, C. B., Venn, S., Goetz, S., Andreu-Hayles, L., Elmendorf, S., Ravolainen, V., Welker, J., Grogan, P., Epstein, H. E., & Hik, D. S. (2011). Shrub expansion in tundra ecosystems: dynamics, impacts and research priorities. *Environmental Research Letters*, *6*(4), 15pp. https://doi.org/10.1088/1748-9326/6/4/045509

Myers, N., Mittermeier, R. A., Mittermeier, C. G., da Fonseca, G. A. B., & Kent, J. (2000). Biodiversity hotspots for conservation priorities. *Nature*, *403*(6772), 853–858. http://doi.org/10.1038/35002501

Nair, P. K. R. (1985). Classification of agroforestry systems. *Agroforestry Systems*, *3*(2), 97–128. http://doi.org/10.1007/BF00122638

Nair, P. K. R., Gordon, A. M., & Mosqueda-Losada, M. R. (2008). Agroforestry. In S. E. Jorgensen & B. Fath (Eds.), *Encyclopedia of Ecology. Vol. 1*. Amsterdam, The Netherlands: Elsevier.

Naito, A. T., & Cairns, D. M. (2015). Patterns of shrub expansion in Alaskan arctic river corridors suggest phase transition. *Ecology and Evolution*, *5*(1), 87–101. http://doi.org/10.1002/ece3.1341

Navarro-González, R., Rainey, F. A., Molina, P., Bagaley, D. R., Hollen, B. J., de la Rosa, J., Small, A. M., Quinn, R. C., Grunthaner, F. J., Cáceres, L., Gomez-Silva, B., & McKay, C. P. (2003). Mars-like soils in the Atacama Desert, Chile, and the dry limit of microbial life. *Science*, *302*(5647), 1018–1021. https://doi.org/10.1126/science.1089143

Naylor, R. L. (2006). Environmental safeguards for open-ocean aquaculture. *Issues in Science and Technology*, *22*(3), 53–58.

Neff, J., Reynolds, R., Belnap, J., & Lamothe, P. (2005). Multi-decadal impacts of grazing on soil physical and biogeochemical properties in southeast Utah. *Ecological Applications*, *15*, 87–95. http://doi.org/10.1890/04-0268

Nelson, A., & Chomitz, K. M. (2011). Effectiveness of strict vs. multiple use protected areas in reducing tropical forest fires: a global analysis using matching methods. *PLoS ONE*, *6*(8), e22722. http://doi.org/10.1371/journal.pone.0022722

Nelson, E., Helmus, M. R., Cavender-Bares, J., Polasky, S., Lasky, J. R., Zanne, A. E., Pearse, W. D., Kraft, N. J. B., Miteva, D. A., Fagan, W. F., Miteva, D., & Fagan, W. (2016). Commercial plant production and consumption still follow the latitudinal gradient in species diversity despite economic globalization. *PLoS ONE*, *11*(10), e0163002. https://doi.org/https://doi.org/10.1371/journal.pone.0163002

Nelson, M. D., Flather, C. H., Riitters, K. H., Sieg, C., & Garner, J. D. (2016). National report on sustainable forests—2015: Conservation of biological diversity. In S. M. Stanton & G. A. Christensen (Eds.), *Pushing Boundaries: New Directions in Inventory Techniques and Applications: Forest Inventory and Analysis (FIA) Symposium 2015* (p. 375). Portland, USA: United States Department of Agriculture. http://doi.org/10.2737/PNW-GTR-931

Nemogá Soto, G. R. (2011). Conocimientos tradicionales, biodiversidad y derechos de propiedad intelectual: Necesidad de una perspectiva de análisis propia. In G. R. Nemogá (Ed.), *Naciones Indígenas en los Estados Contemporáneos* (pp. 231–252). Bogotá, Colombia: Universidad Nacional de Colombia.

Nepstad, D., McGrath, D., Stickler, C., Alencar, A., Azevedo, A., Swette, B., Bezerra, T., DiGiano, M., Shimada, J., Seroa da Motta, R., Armijo, E., Castello, L., Brando, P., Hansen, M. C., McGrath-Horn, M., Carvalho, O., & Hess, L. (2014). Slowing Amazon deforestation through public policy and interventions in beef and soy supply chains. *Science*, *344*(6188), 1118–1123.https://doi.org/10.1126/science.1248525

Nepstad, D., Schwartzman, S., Bamberger, B., Santilli, M., Ray, D., Schlesinger, P., Lefebvre, P., Alencar, A., Prinz, E., Fiske, G., & Rolla, A. (2006). Inhibition of Amazon deforestation and fire by parks and indigenous lands. *Conservation Biology*, *20*(1), 65–73. https://doi.org/10.1111/j.1523-1739.2006.00351.x

Neubauer, P., Jensen, O. P., Hutchings, J. A., & Baum, J. K. (2013). Resilience and recovery of overexploited marine populations. *Science*, *340*(6130), 347–349. http://doi.org/10.1126/science.1230441

Niemeyer, H. M. (1995). Biologically active compounds from Chilean medicinal plants. In J. T. Arnason, R. Matta, & J. T. Romeo (Eds.), *Recent Advances in Phytochemistry. Phytochemistry of Medicinal Plants, Vol. 29* (pp. 137–159). New York, USA: Springer. http://doi.org/10.1007/978-1-4899-1778-2_7

Nipperess, D., & Wilson, P. (2017). PDcalc: An implementation of the Phylogenetic Diversity (PD) calculus in R. R package version 0.3.0.9000. Retrieved from https://github.com/davidnipperess/PDcalc

Nobles, T., & Zhang, Y. (2011). Biodiversity loss in freshwater mussels: Importance, threats, and solutions. In O. Grillo & G. Venora (Eds.), *Biodiversity Loss in a Changing Planet* (pp. 137–162). Open Access: Intech.http://doi.org/10.5772/25102

Nogueira, C., Colli, G. R., & Martins, M. (2009). Local richness and distribution of the lizard fauna in natural habitat mosaics of the Brazilian Cerrado. *Austral Ecology*, *34*(1), 83–96.

http://doi.org/10.1111/j.1442-9993.2008.01887.x
Norambuena, H., Escobar, S., & Rodriguez, F. (2000). The biocontrol of Gorse, *Ulex europaeus*, in Chile: a progress report. In N. R. Spencer (Ed.), *Proceedings of the International Symposium on Biological Control of Weeds* (pp. 955–961). Bozeman, Montana, USA: Montana State University. Retrieved from https://www.researchgate.net/profile/Fernando_Rodriguez12/publication/237442332_The_Biocontrol_of_Gorse_Ulex_europaeus_in_Chile_A_Progress_Report/links/575ab24008aec91374a614e5.pdf
Northern Great Plains Program, W. (2016). *Grasslands*. Retrieved from https://www.worldwildlife.org/habitats/grasslands
Noss, R., LaRoe, E., & Scott, J. (1995). *Endangered Ecosystems of the United States: A Preliminary Assessment of Loss and Degradation*. Washington D.C., USA: US Department of the Interior. National Biological Service.
Nowak, D., Hirabayashi, S., Bodine, A., & Hoehn, R. (2013). Modeled PM2.5 removal by trees in ten U.S. cities and associated health effects. *Environmental Pollution*, *178*, 395–402. http://doi.org/10.1016/j.envpol.2013.03.050
Núñez, L., Grosjean, M., & Cartajena, I. (2002). Human occupations and climate change in the Puna de Atacama, Chile. *Science*, *298*(5594), 821–824. http://doi.org/10.1126/science.1076449
Nuzzo, V. A. (1986). Extent and status of midwest oak savanna: Presettlement and 1985. *Natural Areas Journal*, *6*(2), 6–36.
O'Connor, M. I., Gonzalez, A., Byrnes, J. E. K., Cardinale, B. J., Duffy, J. E., Gamfeldt, L., Griffin, J. N., Hooper, D., Hungate, B. A., Paquette, A., Thompson, P. L., Dee, L. E., & Dolan, K. L. (2017). A general biodiversity–function relationship is mediated by trophic level. *Oikos*, *126*(1), 18–31. https://doi.org/10.1111/oik.03652
O'Donnell, J. A., Romanovsky, V. E., Harden, J. W., & McGuire, A. D. (2009). The effect of moisture content on the thermal conductivity of moss and organic soil horizons from black spruce ecosystems in interior Alaska. *Soil Science*, *174*(12), 646–651. http://doi.org/10.1097/SS.0b013e3181c4a7f8
OBIS. (n.d.). No Title. Retrieved from http://www.iobis.org/
Ocampo-Peñuela, N., & Pimm, S. L. (2015). Elevational ranges of montane birds and deforestation in the Western Andes of Colombia. *PLoS ONE*, *10*(12), e0143311. http://doi.org/10.1371/journal.pone.0143311
Ochoa-Tocachi, B. F., Buytaert, W., & De Bièvre, B. (2016). Regionalization of land-use impacts on streamflow using a network of paired catchments. *Water Resources Research*, *52*(9), 6710–6729. http://doi.org/10.1002/2016WR018596
OECD/ECLAC. (2016). *OECD Environmental Performance Reviews: Chile 2016* (OECD Environmental Performance Reviews). Paris, France: OECD Publishing. Retrieved from http://www.oecd-ilibrary.org/environment/oecd-environmental-performance-reviews-chile-2016_9789264252615-en
Oechel, W. C., Vourlitis, G. L., Hastings, S. J., Zulueta, R. C., Hinzman, L., & Kane, D. (2000). Acclimation of ecosystem CO2 exchange in the Alaskan Arctic in response to decadal climate warming. *Nature*, *406*(6799), 978–981. http://doi.org/10.1038/35023137
Ojeda, R. A., Stadler, J., & Brandl, R. (2003). Diversity of mammals in the tropical-temperate Neotropics: hotspots on a regional scale. *Biodiversity and Conservation*, *12*(7), 1431–1444. http://doi.org/10.1023/A:1023625125032
Oliveira, J. A. (2003). Diversidade de mamíferos e o estabelecimento de áreas prioritárias para a conservação do bioma Caatinga. In J. M. C. da. Silva, M. Tabarelli, T. M. Fonseca, & L. V. Lins (Eds.), *Biodiversidade da Caatinga: Áreas e Ações Prioritárias para a Conservação* (pp. 263–282). Brasilia D.F., Brazil: Ministério do Meio Ambiente/Universidade Federal de Pernambuco.
Oliveira, M. D., Campos, M. C. S., Paolucci, E. M., Mansur, M. C. D., & Hamilton, S. K. (2015). Colonization and spread of *Limnoperna fortunei* in South America. In D. Boltovskoy (Ed.), *Limnoperna fortunei: the Ecology, Distribution and Control of a Swiftly Spreading Invasive Fouling Mussel* (pp. 333–355). Springer International Publishing. http://doi.org/10.1007/978-3-319-13494-9_19
Oliveira, P. T. S., Nearing, M. A., & Wendland, E. (2015). Orders of magnitude increase in soil erosion

associated with land use change from native to cultivated vegetation in a Brazilian savannah environment. *Earth Surface Processes and Landforms*, *40*(11), 1524–1532. http://doi.org/10.1002/esp.3738

Omaeqnomenew Masenahekan. (2004). *Menominee Indian Tribe of Wisconsin. Facts and Figures Reference* (3rd ed.). Menominee Indian Tribe of Wisconsin Department of Administration Community Resource Planner – Brian Kowalkowski. Retrieved from https://www.menominee-nsn.gov/CulturePages/Documents/FactsFigureswithSupplement.pdf

Orensanz, J. M., Schwindt, E., Pastorino, G., Bortolus, A., Casas, G., Darrigran, G., Elías, R., López Gappa, J. J., Obenat, S., Pascual, M., Penchaszadeh, P., Piriz, M. L., Scarabino, F., Spivak, E. D., & Vallarino, E. A. (2002). No longer the pristine confines of the world ocean: a survey of exotic marine species in the southwestern Atlantic. *Biological Invasions*, *4*(1–2), 115–143. https://doi.org/10.1023/A:1020596916153

Ortega-Álvarez, R., & MacGregor-Fors, I. (2011). Spreading the word: The ecology of urban birds outside the United States, Canada, and western Europe. *The Auk: Ornithological Advances*, *128*(2), 415–418. http://doi.org/10.1525/auk.2011.10082

Ortega-Baes, P., & Godínez-Alvarez, H. (2006). Global diversity and conservation priorities in the Cactaceae. *Biodiversity and Conservation*, *15*(3), 817–827. http://doi.org/10.1007/s10531-004-1461-x

Orwig, D. A., Foster, D. R., & Mausel, D. L. (2002). Landscape patterns of hemlock decline in New England due to the introduced hemlock woolly adelgid. *Journal of Biogeography*, *29*(10–11), 1475–1487. http://doi.org/10.1046/j.1365-2699.2002.00765.x

Overbeck, G. E., Vélez-Martin, E., Scarano, F. R., Lewinsohn, T. M., Fonseca, C. R., Meyer, S. T., Muller, S. C., Ceotto, P., Dadalt, L., Durigan, G., Ganade, G., Gossner, M. M., Guadagnin, D. L., Lorenzen, K., Jacobi, C. M., Weisser, W. W., & Pillar, V. D. (2015). Conservation in Brazil needs to include non-forest ecosystems. *Diversity and Distributions*, *21*(12), 1455–1460. https://doi.org/10.1111/ddi.12380

Pacheco, P., Aguilar-Støen, M., Börner, J., Etter, A., Putzel, L., & Vera Diaz, M. del C. (2011). Landscape transformation in tropical Latin America: assessing trends and policy implications for REDD+. *Forests*, *2*(1), 1–29. http://doi.org/10.3390/f2010001

Pagad, S., Hayes, K., Katsanevakis, S., & Costello, M. J. (2017). World Register of Introduced Marine Species (WRIMS). Retrieved March 15, 2017, from http://www.marinespecies.org/introduced

Paglia, A. P., da Fonseca, G. A., Rylands, A. B., Herrmann, G., Aguiar, L. M., Chiarello, A. G., Leite, Y. L., Costa, L. P., Siciliano, S., Kierulff, M. C. M., Mendes, S. L., Tavares, V., Mittermeier, R. A., & Patton, J. L. (2012). *Lista Anotada dos Mamíferos do Brasil/Annotated Checklist of Brazilian Mammals*. *Occasional Papers in Conservation Biology N°6* (2nd ed.). Arlington, USA: Conservation International.

Painter, T. H., Deems, J. S., Belnap, J., Hamlet, A. F., Landry, C. C., & Udall, B. (2010). Response of Colorado River runoff to dust radiative forcing in snow. *PNAS*, *107*(40), 17125–17130. http://doi.org/10.1073/pnas.0913139107

Paquette, A., & Messier, C. (2011). The effect of biodiversity on tree productivity: from temperate to boreal forests. *Global Ecology and Biogeography*, *20*(1), 170–180. http://doi.org/10.1111/j.1466-8238.2010.00592.x

Parés-Ramos, I. K., Gould, W. A., & Aide, T. M. (2008). Agricultural abandonment, suburban growth, and forest expansion in Puerto Rico between 1991 and 2000. *Ecology and Society*, *13*(2).

Paritsis, J., & Aizen, M. A. (2008). Effects of exotic conifer plantations on the biodiversity of understory plants, epigeal beetles and birds in *Nothofagus dombeyi* forests. *Forest Ecology and Management*, *255*(5–6), 1575–1583. http://doi.org/10.1016/j.foreco.2007.11.015

Parra, F., & Casas, A. (2016). Origen y difusión de la domesticación y la agricultura en el Nuevo Mundo. In A. Casas, J. Torres-Guevara, & F. Parra (Eds.), *Domesticación en el Continente Americano Vol. 1. Investigación Manejo de Biodiversidad y Evolución Dirigida por las Culturas del Nuevo Mundo* (pp. 159–188). México D.F., México and Lima, Perú: Universidad Nacional Autónoma de México /Universidad Nacional Agraria La Molina.

Paruelo, J. M., Garbulsky, M. F., Guerschman, J. P., & Jobbágy, E. G. (2004). Two decades of normalized difference vegetation index changes in South America: identifying the imprint of global change.

International Journal of Remote Sensing, *25*(14), 2793–2806. http://doi.org/10.1080/01431160310001619526

Pauchard, A., Aguayo, M., Peña, E., & Urrutia, R. (2006). Multiple effects of urbanization on the biodiversity of developing countries: the case of a fast-growing metropolitan area (Concepción, Chile). *Biological Conservation*, *127*(3), 272–281. http://doi.org/10.1016/j.biocon.2005.05.015

Pauchard, A., & Barbosa, O. (2013). Regional assessment of Latin America: Rapid urban development and social economic inequity threaten biodiversity hotspots. In T. Elmqvist, M. Fragkias, J. Goodness, B. Güneralp, P. J. Marcotullio, R. I. McDonald, … C. Wilkinson (Eds.), *Urbanization, Biodiversity and Ecosystem Services: Challenges and Opportunities: A Global Assessment* (pp. 589–608). New York, USA: Springer.

Pauchard, A., García, R. A., Peña, E., González, C., Cavieres, L. A., & Bustamante, R. O. (2008). Positive feedbacks between plant invasions and fire regimes: *Teline monspessulana* (L.) K. Koch (Fabaceae) in central Chile. *Biological Invasions*, *10*(4), 547–553. http://doi.org/10.1007/s10530-007-9151-8

Pauchard, A., Kueffer, C., Dietz, H., Daehler, C. C., Alexander, J., Edwards, P. J., Ramon Arevalo, J., Cavieres, L. A., Guisan, A., Haider, S., Jakobs, G., McDougall, K., Millar, C. I., Naylor, B. J., Parks, C. G., Rew, L. J., & Seipel, T. (2009). Ain't no mountain high enough: plant invasions reaching new elevations. *Frontiers in Ecology and the Environment*, *7*(9), 479–486. https://doi.org/10.1890/080072

Paul, M. J., & Meyer, J. L. (2001). Streams in the urban landscape. *Annual Review of Ecology and Systematics*, *32*(2001), 333–365. http://doi.org/10.1146/annurev.ecolsys.32.081501.114040

Paulich, N. (2010). Increasing private conservation through incentive mechanisms. *Journal of Animal Law and Policy*, *3*, 106–158.

Pauro, J., Gonzáles, M., Gamarra C, B., Pauro R, J., Mamani M, F., & Huerta, R. (2011). Plantas alimenticias, medicinales y biocidas de la comuna de Muñani y Suatia, provincia de Lampa (Puno-Perú). *Ecología Aplicada*, *10*(1), 41–49.

Pavez, E. F., Lobos, G. A., & Jaksic, F. M. (2010). Long-term changes in landscape and in small mammal and raptor assemblages in central Chile. *Revista Chilena de Historia Natural*, *83*(1), 99–111.

Pavlik, B. M., & Skinner, M. W. (1994). Ecological characteristics of California's rare plants. In *Inventory of Rare and Endangered Vascular Plants of California* (5th ed., pp. 4–6). Sacramento, California: California Native Plant Society.

Pearse, W. D., Cavender-Bares, J., Hobbie, S. E., Avolio, M. L., Bettez, N., Roy Chowdhury, R., Darling, L. E., Groffman, P. M., Grove, J. M., Hall, S. J., Heffernan, J. B., Learned, J., Neill, C., Nelson, K. C., Pataki, D. E., Ruddell, B. L., Steele, M. K., & Trammell, T. L. E. (2018). Homogenization of plant diversity, composition, and structure in North American urban yards. *Ecosphere*, *9*(2), e02105. https://doi.org/10.1002/ecs2.2105

Pedersen, A. B., Jones, K. E., Nunn, C. L., & Altizer, S. (2007). Infectious diseases and extinction risk in wild mammals. *Conservation Biology*, *21*(5), 1269–1279. http://doi.org/10.1111/j.1523-1739.2007.00776.x

Peixoto, R. B., Marotta, H., Bastviken, D., & Enrich-Prast, A. (2016). Floating aquatic macrophytes can substantially offset open water CO2 emissions from tropical floodplain lake ecosystems. *Ecosystems*, *19*(4), 724–736. http://doi.org/10.1007/s10021-016-9964-3

Pell, A. N. (1999). Animals and agroforestry in the tropics. In L. E. Buck, J. P. Lassoie, & E. C. M. Fernandes (Eds.), *Agroforestry in Sustainable Agricultural Systems* (pp. 33–46). Boca Raton, USA: CRC Press and Lewis Publishers.

Peña-Gómez, F. T., Guerrero, P. C., Bizama, G., Duarte, M., & Bustamante, R. O. (2014). Climatic niche conservatism and biogeographical non-equilibrium in *Eschscholzia californica* (Papaveraceae), an invasive plant in the Chilean Mediterranean region. *PLoS ONE*, *9*(8), e105025. http://doi.org/10.1371/journal.pone.0105025

Peña, E., Hidalgo, M., Langdon, B., & Pauchard., A. (2008). Patterns of spread of Pinus contorta Dougl. ex Loud. invasion in a Natural Reserve in southern South America. *Forest Ecology and Management*, *256*(5), 1049–1054. http://doi.org/10.1016/j.foreco.2008.06.020

Peralvo, M. F., Cuesta, F., & van Manen, F. (2005). Delineating priority habitat areas for the conservation of Andean bears in northern Ecuador. *Ursus*, *16*(2), 222–233. http://doi.org/10.2192/1537-6176(2005)016%5B0222:DPHAFT%5D2.0.CO;2

Pereira-Garbero, R., Barreneche, J. M., Laufer, G., Achaval, F., & Arim, M. (2013). Mamíferos invasores en Uruguay, historia, perspectivas y consecuencias. *Revista Chilena de Historia Natural*, *86*(4), 403–421. http://doi.org/10.4067/S0716-078X2013000400003

Perepelizin, P. V., & Faggi, A. M. (2009). Diversidad de aves en tres barrios de la ciudad de Buenos Aires, Argentina. *Multequina*, *18*(2), 71–85.

Peres, C. A., Emilio, T., Schietti, J., Desmoulière, S. J. M., & Levi, T. (2016). Dispersal limitation induces long-term biomass collapse in overhunted Amazonian forests. *PNAS*, *113*(4), 892–897.http://doi.org/10.1073/pnas.1516525113

Pérez-Cruz, Y. G., Rangel-Ruiz, L. J., & Gamboa-Aguilar, J. (2013). Metals in clams and sediments in the marshes of Centla Biosphere Reserve, Tabasco, Mexico. *Hidrobiologica*, *23*(1), 1–8.

Pérez-Ruzafa, A., Alvarado, J. J., Solís-Marín, F. A., Hernández, J. C., Morata, A., Marcos, C., Abreu-Pérez, M., Aguilera, O., Alió, J., Bacallado-Aránega, J. J., Barraza, E., Benavides-Serrato, M., Benítez-Villalobos, F., Betancourt-Fernández, L., Borges, M., Brandt, M., Brogger, M. I., Borrero-Pérez, G. H., Buitrón-Sánchez, B. E., Campos, L. S., Cantera, J. R., Clemente, S., Cohen-Renfijo, M., Coppard, S. E., Costa-Lotufo, L. V., Del Valle-García, R., Díaz De Vivar, M. E., Díaz-Martínez, J. P., Díaz, Y., Durán-González, A., Epherra, L., Escolar, M., Francisco, V., Freire, C. A., García-Arrarás, J. E., Gil, D. G., Guarderas, P., Hadel, V. F., Hearn, A., Hernández-Delgado, E. A., Herrera-Moreno, A., Herrero-Pérezrul, M. D., Hooker, Y., Honey-Escandón, M. B. I., Lodeiros, C., Luzuriaga, M., Manso, C. L. C., Martín, A., Martínez, M. I., Martínez, S., Moro-Abad, L., Mutschke, E., Navarro, J. C., Neira, R., Noriega, N., Palleiro-Nayar, J. S., Pérez, A. F., Prieto-Ríos, E., Reyes, J., Rodríguez-Barreras, R., Rubilar, T., Sancho-Mejías, T. I., Sangil, C., Silva, J. R. M. C., Sonnenholzner, J. I., Ventura, C. R. R., Tablado, A., Tavares, Y., Tiago, C. G., Tuya, F., & Williams, S. M. (2013). Latin America echinoderm biodiversity and biogeography: patterns and affinities. In J. J. Alvarado & F. A. Solís-Marín (Eds.), *Echinoderm Research and Diversity in Latin America* (pp. 511–542). Berlin, Germany: Springer.

Pérez, L. (2008). Aportes para la comprensión del imaginario Otomí. In C. H. Durand (Ed.), *El Derecho al Desarrollo Social: Una Visión Desde el Multiculturalismo: El Caso de los Pueblos Indígenas* (pp. 273–292). Ciudad de México, México: Editorial Porrúa.

Perfecto, I., & Vandermeer, J. (2008). Biodiversity conservation in tropical agroecosystems. *Annals of the New York Academy of Sciences*, *1134*(1), 173–200. http://doi.org/10.1196/annals.1439.011

Peri, P. L., Dube, F., & Varella, A. C. (2016). Silvopastoral systems in the subtropical and temperate zones of South America: an overview. In P. L. Peri, F. Dube, & A. Varella (Eds.), *Silvopastoral Systems in Southern South America* (pp. 1–8). Cham, Switzerland: Springer.

Peri, P. L., Ladd, B., Pepper, D. A., Bonser, S. P., Laffan, S. W., & Amelung, W. (2012). Carbon (δ 13C) and nitrogen (δ 15N) stable isotope composition in plant and soil in Southern Patagonia's native forests. *Global Change Biology*, *18*(1), 311–321. http://doi.org/10.1111/j.1365-2486.2011.02494.x

Periago, M. E., Chillo, V., & Ojeda, R. A. (2015). Loss of mammalian species from the South American Gran Chaco: empty savanna syndrome? *Mammal Review*, *45*(1), 41–53. http://doi.org/10.1111/mam.12031

Peroni, N., & Hanazaki., N. (2002). Current and lost diversity of cultivated varieties, especially cassava, under swidden cultivation systems in the Brazilian Atlantic Forest. *Agriculture Ecosystems & Environment*, *92*(2), 171–183. http://doi.org/10.1016/S0167-8809(01)00298-5

Perry, L., Dickau, R., Zarrillo, S., Holst, I., Pearsall, D. M., Piperno, D. R., Berman, M. J., Cooke, R. G., Rademaker, K., Ranere, A. J., Raymond, J. S., Sandweiss, D. H., Scaramelli, F., Tarble, K., & Zeidler, J. A. (2007). Starch fossils and the domestication and dispersal of chili peppers (*Capsicum* spp. L.) in the Americas. *Science*, *315*(5814), 986–988. https://doi.org/10.1126/science.1136914

Perry, W. L., Feder, J. L., & Lodge, D. M. (2001). Implications of hybridization between introduced and resident Orconectes crayfishes. *Conservation Biology*, *15*(6), 1656–1666. http://doi.org/10.1046/j.1523-1739.2001.00019.x

Petchey, O. L., & Gaston, K. J. (2002). Functional diversity (FD), species richness and community composition. *Ecology Letters*, *5*(3), 402–411. http://doi.org/10.1046/j.1461-0248.2002.00339.x

Petermann, J. S., Fergus, A. J. F., Turnbull, L. A., & Schmid, B. (2008). Janzen-Connell effects are widespread and strong enough to maintain diversity in grasslands. *Ecology*, *89*(9), 2399–2406.

http://doi.org/10.1890/07-2056.1
Peterson, D. W., & Reich, P. B. (2001). Prescribed fire in oak savanna: fire frequency effects on stand structure and dynamics. *Ecological Applications*, *11*(3), 914–927. http://doi.org/10.1890/1051-0761(2001)011%5B0914:PFIOSF%5D2.0.CO;2
Peterson, D. W., & Reich, P. B. (2008). Fire frequency and tree canopy structure influence plant species diversity in a forest-grassland ecotone. *Plant Ecology*, *194*(1), 5–16. http://doi.org/10.1007/s11258-007-9270-4
Petitpas, R., Ibarra, J. T., Miranda, M., & Bonacic, C. (2016). Spatial patterns over a 24-year period show an increase in native vegetation cover and decreased fragmentation in Andean temperate landscapes, Chile. *Ciencia E Investigación Agraria*, *43*(3), 384–395. http://doi.org/10.4067/S0718-16202016000300005
Philpott, S. M., Arendt, W. J., Armbrecht, I., Bichier, P., Diestch, T. V., Gordon, C., Greenberg, R., Perfecto, I., Reynoso-Santos, R., Soto-Pinto, L., Tejeda-Cruz, C., Williams-Linera, G., Valenzuela, J., & Zolotoff, J. M. (2008). Biodiversity loss in Latin American coffee landscapes: Review of the evidence on ants, birds, and trees. *Conservation Biology*, *22*(5), 1093–1105. https://doi.org/10.1111/j.1523-1739.2008.01029.x
Pimentel, D. (2002). *Biological Invasions – Economic and Environmental Costs of Alien Plant, Animal, and Microbe Species*. (D. Pimentel, Ed.). Boca Raton, USA: CRC Press.
Pimentel, D., Zuniga, R., & Morrison, D. (2005). Update on the environmental and economic costs associated with alien-invasive species in the United States. *Ecological Economics*, *52*(3), 273–288. http://doi.org/10.1016/j.ecolecon.2004.10.002
Pimm, S. L., Jones, H. L., & Diamond, J. (1988). On the risk of extinction. *The American Naturalist*, *132*(6), 757–785. http://doi.org/10.1086/284889
Pincetl, S., Watt, T., & Santos, M. J. (2016). Land use regulation for resource conservation. In H. Mooney & E. Zavaleta (Eds.), *Ecosystem of California* (pp. 899–924). Oakland, USA: University of California Press.
Pincheira-Donoso, D., Harvey, L. P., & Ruta, M. (2015). What defines an adaptive radiation? Macroevolutionary diversification dynamics of an exceptionally species-rich continental lizard radiation. *BMC Evolutionary Biology*, *15*(1), 153. http://doi.org/10.1186/s12862-015-0435-9
Pinsky, M. L., Worm, B., Fogarty, M. J., Sarmiento, J. L., & Levin, S. A. (2013). Marine taxa track local climate velocities. *Science*, *341*(6151), 1239–1242. http://doi.org/10.1126/science.1239352
Piñeiro, G., Paruelo, J. M., & Oesterheld, M. (2006). Potential long-term impacts of livestock introduction on carbon and nitrogen cycling in grasslands of southern South America. *Global Change Biology*, *12*(7), 1267–1284. http://doi.org/10.1111/j.1365-2486.2006.01173.x
Piovia-Scott, J., Pope, K., Worth, S. J., Rosenblum, E. B., Poorten, T., Refsnider, J., … Foley, J. (2015). Correlates of virulence in a frog-killing fungal pathogen: evidence from a California amphibian decline. *The ISME Journal*, *9*(7), 1570–1578. http://doi.org/10.1038/ismej.2014.241
Pizano, C., Barón, A. F., Schuur, E. A. G., Crummer, K. G., & Mack, M. C. (2014). Effects of thermo-erosional disturbance on surface soil carbon and nitrogen dynamics in upland arctic tundra. *Environmental Research Letters*, *9*(7), 75006. http://doi.org/10.1088/1748-9326/9/7/075006
Pizano, C., & García, H. (2014). *El Bosque Seco Tropical de Colombia*. Bogota, D.C.: Instituto de Recursos Biológicos Alexander Von Humboldt.
Pizarro Tapia, R., M., Tapia Cornejo, M., Román Arellano, L., Jordán Díaz, C., & Farías Daza, C. (2006). Coeficientes de escorrentía instantáneos para la cuenca del río Tutuvén, VII Región del Maule, Chile. *Bosque*, *27*(2), 83–91.
Pliscoff, P., & Fuentes-Castillo, T. (2011). Representativeness of terrestrial ecosystems in Chile's protected area system. *Environmental Conservation*, *38*(3), 303–311. http://doi.org/10.1017/S0376892911000208
Poeplau, C., & Don, A. (2015). Carbon sequestration in agricultural soils via cultivation of cover crops – A meta-analysis. *Agriculture Ecosystems & Environment*, *200*, 33–41. http://doi.org/10.1016/j.agee.2014.10.024
Polidoro, B. A., Carpenter, K. E., Collins, L., Duke, N. C., Ellison, A. M., Ellison, J. C., Farnsworth, E. J., Fernando, E. S., Kathiresan, K., Koedam, N. E., Livingstone, S. R., Miyagi, T., Moore, G. E., Vien, N.

N., Ong, J. E., Primavera, J. H., Salmo III, S. G., Sanciangco, J. C., Sukardjo, S., Wang, Y., & Yong, J. W. H. (2010). The loss of species: mangrove extinction risk and geographic areas of global concern. *PLoS ONE*, *5*(4), e10095. https://doi.org/10.1371/journal.pone.0010095

Poloczanska, E. S., Brown, C. J., Sydeman, W. J., Kiessling, W., Schoeman, D. S., Moore, P. J., Brander, K., Bruno, J. F., Buckley, L. B., Burrows, M. T., Duarte, C. M., Halpern, B. S., Holding, J., Kappel, C. . V., O'Connor, M. I., Pandolfi, J. M., Parmesan, C., Schwing, F. B., Thompson, R. C., & Richardson, A. J. (2013). Global imprint of climate change on marine life. *Nature Climate Change*, *3*(10), 919–925. https://doi.org/10.1038/Nclimate1958

Portela, S. I., Andriulo, A. E., Jobbágy, E. G., & Sasal, M. C. (2009). Water and nitrate exchange between cultivated ecosystems and groundwater in the Rolling Pampas. *Agriculture, Ecosystems and Environment*, *134*(3–4), 277–286. http://doi.org/10.1016/j.agee.2009.08.001

Portela, S. I., Andriulo, A. E., Sasal, M. C., Mary, B., & Jobbágy, E. G. (2006). Fertilizer vs. organic matter contributions to nitrogen leaching in cropping systems of the Pampas: 15N application in field lysimeters. *Plant and Soil*, *289*(1–2), 265–277. http://doi.org/10.1007/s11104-006-9134-z

Portillo-Quintero, C. A., & Sánchez-Azofeifa, G. A. (2010). Extent and conservation of tropical dry forests in the Americas. *Biological Conservation*, *143*(1), 144–155. http://doi.org/10.1016/j.biocon.2009.09.020

Posey, D. A. (1999). *Cultural and Spiritual Values of Biodiversity*. London, UK: UNEP and Intermediate Technology Publications.

Pott, A., Oliveira, A. K. M., Damasceno-Junior, G. A., & Silva, J. S. V. (2011). Plant diversity of the Pantanal wetland. *Brazilian Journal of Biology*, *71*(1), 265–273. http://doi.org/10.1590/S1519-69842011000200005

Potts, S. G., Biesmeijer, J. C., Kremen, C., Neumann, P., Schweiger, O., & Kunin, W. E. (2010). Global pollinator declines: trends, impacts and drivers. *Trends in Ecology and Evolution*, *25*(6), 345–353. http://doi.org/10.1016/j.tree.2010.01.007

Price, S. J., Cecala, K. K., Browne, R. A., & Dorcas, M. E. (2011). Effects of urbanization on occupancy of stream salamanders. *Conservation Biology*, *25*(3), 547–555. http://doi.org/10.1111/j.1523-1739.2010.01627.x

Prival, D. B., & Schroff, M. J. (2012). A 13-year study of a northern population of twin-spotted rattlesnakes (*Crotalus pricei*): growth, reproduction, survival, and conservation. *Herpetological Monographs*, *26*(1), 1–18.

Programa Estado de la Nacion. (2008). *Estado de la Región en Desarrollo Humano Sostenible: Un Informe desde Centroamérica y para Centroamérica*. San José, Costa Rica: Estado de la Nación.

Pyron, M., Becker, J. C., Broadway, K. J., Etchison, L., Minder, M., DeColibus, D., Chezem, M., Wyatt, K. H., & Murry, B. A. (2017). Are long-term fish assemblage changes in a large US river related to the Asian Carp invasion? Test of the hostile take-over and opportunistic dispersal hypotheses. *Aquatic Sciences*, *79*(3), 631–642. https://doi.org/10.1007/s00027-017-0525-4

Pyron, R. A. (2014). Biogeographic analysis reveals ancient continental vicariance and recent oceanic dispersal in amphibians. *Systematic Biology*, *63*(5), 779–797. http://doi.org/10.1093/sysbio/syu042

Quijas, S., Schmid, B., & Balvanera, P. (2010). Plant diversity enhances provision of ecosystem services: a new synthesis. *Basic and Applied Ecology*, *11*(7), 582–593. http://doi.org/10.1016/j.baae.2010.06.009

Quinn, P. K., Shaw, G., Andrews, E., Dutton, E. G., Ruoho-Airola, T., & Gong, S. L. (2007). Arctic haze: current trends and knowledge gaps. *Tellus, Series B: Chemical and Physical Meteorology*, *59*(1), 99–114. http://doi.org/10.1111/j.1600-0889.2006.00238.x

Rabatel, A., Francou, B., Soruco, A., Gomez, J., Cáceres, B., Ceballos, J. L., Basantes, R., Vuille, M., Sicart, J.-E., Huggel, C., Scheel, M., Lejeune, Y., Arnaud, Y., Collet, M., Condom, T., Consoli, G., Favier, V., Jomelli, V., Galarraga, R., Ginot, P., Maisincho, L., Mendoza, J., Ménégoz, M., Ramirez, E., Ribstein, P., Suarez, W., Villacis, M., & Wagnon, P. (2013). Current state of glaciers in the tropical Andes: a multi-century perspective on glacier evolution and climate change. *The Cryosphere*, *7*(1), 81–102. https://doi.org/10.5194/tc-7-81-2013

Rademaker, K., Hodgins, G., Moore, K., Zarrillo, S., Miller, C., Bromley, G. R. M., Leach, P., Reid, D. A., Álvarez,

W. Y., & Sandweiss, D. H. (2014). Paleoindian settlement of the high-altitude Peruvian Andes. *Science*, *346*(6208), 466–469. https://doi.org/10.1126/science.1258260

R Development Core Team. (2017). *R: A Language and Environment for Statistical Computing*. Vienna, Austria: R Foundation for Statistical Computing. Retrieved from https://cran.r-project.org/

RAISG. (2015). *Deforestación en la Amazonía (1970-2013)*. São Paulo: Instituto Socioambiental. Retrieved from www.raisg.socioambiental.org

Ramankutty, N., Evan, A. T., Monfreda, C., & Foley, J. A. (2010). Global Agricultural Lands: Pastures, 2000 Palisades, NY: NASA Socioeconomic Data and Applications Center (SEDAC). Retrieved fromhttp://sedac.ciesin.columbia.edu/data/set/aglands-pastures-2000

Ramankutty, N., Foley, J. A., & Olejniczak, N. J. (2002). People on the land: changes in global population and croplands during the 20th century. *Ambio*, *31*(3), 251–257. http://doi.org/10.1579/0044-7447-31.3.251

Ramjohn, I. A., Murphy, P. G., Burton, T. M., & Lugo, A. E. (2012). Survival and rebound of Antillean dry forests: role of forest fragments. *Forest Ecology and Management*, *284*, 124–132. http://doi.org/10.1016/j.foreco.2012.08.001

Ramos, M. A., & Albuquerque, U. P. (2012). The domestic use of firewood in rural communities of the Caatinga: how seasonality interferes with patterns of firewood collection. *Biomass and Bioenergy*, *39*, 147–158. http://doi.org/10.1016/j.biombioe.2012.01.003

Ramos, R. S., Hilgert, N. I., & Lambaré, D. A. (2013). Traditional agriculture and the richness of maize (Zea mays). A case study in Caspalá, Jujuy province, Argentina. *Boletin de La Sociedad Argentina de Botánica*, *48*(3–4), 607–621.

Ratter, J., Ribeiro, J. F., & Bridgewater, S. (1997). The Brazilian cerrado vegetation and threats to its biodiversity. *Annals of Botany*, *80*(3), 223–230. http://doi.org/10.1006/anbo.1997.0469

Raupp, M. J., Shrewsbury, P. M., & Herms, D. A. (2010). Ecology of herbivorous arthropods in urban landscapes. *Annual Review of Entomology*, *55*(1), 19–38. http://doi.org/10.1146/annurev-ento-112408-085351

Rawlins, M. A., Steele, M., Holland, M. M., Adam, J. C., Cherry, J. E., Francis, J. A., Groisman, P. Y. A., Hinzman, L. D., Huntington, T. G., Kane, D. L., Kimball, J. S., Kwok, R., Lammers, R. B., Lee, C. M., Lettenmaier, D. P., Mcdonald, K. C., Podest, E., Pundsack, J. W., Rudels, B., Serreze, M. C., Shiklomanov, A., Skagseth, Ø., Troy, T. J., Vörösmarty, C. J., Wensnahan, M., Wood, E. F., Woodgate, R., Yang, D., Zhang, K., & Zhang, T. (2010). Analysis of the Arctic system for freshwater cycle intensification: observations and expectations. *Journal of Climate*, *23*(21), 5715–5737. https://doi.org/10.1175/2010JCLI3421.1

Ray, D. K., Welch, R. M., Lawton, R. O., & Nair, U. S. (2006). Dry season clouds and rainfall in northern Central America: implications for the Mesoamerican Biological Corridor. *Global and Planetary Change*, *54*(1–2), 150–162. http://doi.org/10.1016/j.gloplacha.2005.09.004

Ray, G. L. (2005). *Invasive Animal Species in Marine and Estuarine Environments: Biology and Ecology*. Vicksburg, USA: US Army Corps of Engineers.

Raymond, P. A., Hartmann, J., Lauerwald, R., Sobek, S., McDonald, C., Hoover, M., Butman, D., Striegl, R., Mayorga, E., Humborg, C., Kortelainen, P., Durr, H., Meybeck, M., Ciais, P., & Guth, P. (2013). Global carbon dioxide emissions from inland waters. *Nature*, *503*(7476), 355–359. https://doi.org/10.1038/nature12760

Raynolds, M. K., Walker, D. A., Ambrosius, K. J., Brown, J., Everett, K. R., Kanevskiy, M., Kofinas, G. P., Romanovsky, V. E., Shur, Y., & Webber, P. J. (2014). Cumulative geoecological effects of 62 years of infrastructure and climate change in ice-rich permafrost landscapes, Prudhoe Bay Oilfield, Alaska. *Global Change Biology*, *20*(4), 1211–1224. https://doi.org/10.1111/gcb.12500

Read, A. J. (2008). The looming crisis: interactions between marine mammals and fisheries. *Journal of Mammalogy*, *89*(3), 541–548. http://doi.org/10.1644/07-MAMM-S-315R1.1

Redfield, A. C. (1934). On the proportions of organic derivatives in sea water and their relation to the composition of plankton. In *James Johnstone Memorial Volume* (pp. 176–192). Liverpool, UK: University Press of Liverpool.

Redman, C. L., Grove, J. M., & Kuby, L. H. (2004). Integrating social science into the long-term ecological research (LTER) network: social dimensions of ecological change and ecological dimensions of social change. *Ecosystems*, *7*(2), 161–171. http://doi.org/10.1007/s10021-003-0215-z

Rehm, E. M., & Feeley, K. J. (2013). Forest patches and the upward migration of timberline in the southern Peruvian Andes. *Forest Ecology and Management*, *305*, 204–211. http://doi.org/10.1016/j.foreco.2013.05.041

Rehm, E. M., & Feeley, K. J. (2015). The inability of tropical cloud forest species to invade grasslands above treeline during climate change: Potential explanations and consequences. *Ecography*, *38*(12), 1167–1175. http://doi.org/10.1111/ecog.01050

Rehm, E. M., & Feeley, K. J. (2016). Seedling transplants reveal species-specific responses of high-elevation tropical treeline trees to climate change. *Oecologia*, *181*(4), 1233–1242. http://doi.org/10.1007/s00442-016-3619-0

Reich, P. B., Tilman, D., Isbell, F., Mueller, K., Hobbie, S. E., Flynn, D. F. B., & Eisenhauer, N. (2012). Impacts of biodiversity loss escalate through time as redundancy fades. *Science*, *336*(6081), 589–592. http://doi.org/10.1126/science.1217909

Reich, P. B., Tilman, D., Naeem, S., Ellsworth, D. S., Knops, J., Craine, J., Wedin, D., & Trost, J. (2004). Species and functional group diversity independently influence biomass accumulation and its response to CO2 and N. *PNAS*, *101*(27), 10101–10106. https://doi.org/10.1073/pnas.0306602101

Reichard, S. H., & White, P. (2001). Horticulture as a pathway of invasive plant introductions in the United States most invasive plants have been introduced for horticultural use by nurseries, botanical gardens, and individuals. *BioScience*, *51*(2), 103–113. http://doi.org/10.1641/0006-3568(2001)051%5B0103:HAAPOI%5D2.0.CO;2

Reis, R. E., Albert, J. S., Di Dario, F., Mincarone, M. M., Petry, P., & Rocha, L. A. (2016). Fish biodiversity and conservation in South America. *Journal of Fish Biology*, *89*(1), 12–47. http://doi.org/10.1111/jfb.13016

Rejmánek, M. (1996). Species richness and resistance to invasions. In G. Orians, R. Dirzo, & J. H. Cushman (Eds.), *Biodiversity and Ecosystem Processes in Tropical Forests. Ecological Studies (Analysis and Synthesis)* (Vol. 122, pp. 153–172). Berlin, Germany: Springer Verlag.

Rejmánek, M. (2017). Vascular plant extinctions in California: a critical assessment. *Diversity and Distributions*, *24*(1), 129–136. http://doi.org/10.1111/ddi.12665

Rennert, K. J., Roe, G., Putkonen, J., & Bitz, C. M. (2009). Soil thermal and ecological impacts of rain on snow events in the circumpolar Arctic. *Journal of Climate*, *22*(9), 2302–2315. http://doi.org/10.1175/2008JCLI2117.1

Resner, K., Yoo, K., Sebestyen, S. D., Aufdenkampe, A., Hale, C., Lyttle, A., & Blum, A. (2015). Invasive earthworms deplete key soil inorganic nutrients (Ca, Mg, K, and P) in a northern hardwood forest. *Ecosystems*, *18*(1), 89–102. http://doi.org/10.1007/s10021-014-9814-0

Revenga, C., & Kura, Y. (2003). *Status and Trends of Biodiversity of Inland Water Ecosystems*. *Technical Series No. 11*. Montreal, Canada: Secretariat of the Convention on Biological Diversity.

Reyes, S., & Figueroa, I. M. (2010). Distribución, superficie y accesibilidad de las áreas verdes en Santiago de Chile. *EURE Revista Latinoamericana de Estudios Urbanos Regionales*, *36*(109), 89–110. http://doi.org/10.4067/S0250-71612010000300004

Rezende, V. L. (2017). Tree species distribution and phylogenetic diversity across southern South America. *Frontiers of Biogeography*, *9*(2), 0–6. http://doi.org/10.21425/F59232082

Ribeiro, M. C., Martensen, A. C., Metzger, J. P., Tabarelli, M., Scarano, F., & Fortin, M. J. (2011). The Brazilian Atlantic Forest: a shrinking biodiversity hotspot. In F. Zachos & J. Habel (Eds.), *Biodiversity Hotspots: Distribution and Protection of Conservation Priority Areas* (pp. 405–434). Berlin, Germany: Springer.

Ricciardi, A. (2006). Patterns of invasion in the Laurentian Great Lakes in relation to changes in vector activity. *Diversity and Distributions*, *12*(4), 425–433. http://doi.org/10.1111/j.1366-9516.2006.00262.x

Richardson, D. M., Williams, P. A., & Hobbs, R. J. (1994). Pine invasions in the southern hemisphere: determinants of spread and invadability. *Journal of Biogeography*, *21*(5), 511–527.

Ricketts, T. H., Dinerstein, E., Boucher, T., Brooks, T. M., Butchart, S. H. M., Hoffmann, M., Lamoreux, J. F., Morrison, J., Parr, M., Pilgrim, J. D., Rodrigues, A. S. L., Sechrest, W., Wallace, G. E., Berlin, K., Bielby, J., Burgess, N. D., Church, D. R., Cox, N., Knox, D., Loucks, C., Luck, G. W., Master, L. L., Moore, R., Naidoo, R., Ridgely, R., Schatz, G. E., Shire, G., Strand, H., Wettengel, W., & Wikramanayake, E. (2005). Pinpointing and preventing imminent extinctions. *PNAS*, *102*(51), 18497–18501. https://doi.org/10.1073/pnas.0509060102

Ricketts, T. H., Regetz, J., Steffan-Dewenter, I., Cunningham, S. A., Kremen, C., Bogdanski, A., Gemmill-Herren, B., Greenleaf, S. S., Klein, A. M., Mayfield, M. M., Morandin, L. A., Ochieng', A., & Viana, B. F. (2008). Landscape effects on crop pollination services: are there general patterns? *Ecology Letters*, *11*(5), 499–515. https://doi.org/10.1111/j.1461-0248.2008.01157.x

Rimski-Korsakov, H., Zubillaga, M. S., Landriscini, M. R., & Lavado, R. S. (2016). Maize and cover crop sequence in the Pampas: effect of fertilization and water stress on the fate of nitrogen. *Journal of Soil and Water Conservation*, *71*(1), 12–20. http://doi.org/10.2489/jswc.71.1.12

Roberts, P., Hunt, C., Arroyo-Kalin, M., Evans, D., & Boivin, N. (2017). The deep human prehistory of global tropical forests and its relevance for modern conservation. *Nature Plants*, *3*(8), 17093. http://doi.org/10.1038/nplants.2017.93

Robinson, R. A., Lawson, B., Toms, M. P., Peck, K. M., Kirkwood, J. K., Chantrey, J., Clatworthy, I. R., Evans, A. D., Hughes, L. A., Hutchinson, O. C., John, S. K., Pennycott, T. W., Perkins, M. W., Rowley, P. S., Simpson, V. R., Tyler, K. M., & Cunningham, A. A. (2010). Emerging infectious disease leads to rapid population declines of common british birds. *PLoS ONE*, *5*(8), e12215. https://doi.org/10.1371/journal.pone.0012215

Robson, J., & Berkes, F. (2012). Sacred nature and community conserved areas. In S. Pilgrim & J. Pretty (Eds.), *Nature and Culture: Rebuilding Lost Connections* (pp. 197–216). London, UK and Washington DC., USA: Earthscan.

Rodewald, P. (Ed.). (2015). *The Birds of North America*. Ithaca, USA: Cornell Laboratory of Ornithology. Retrieved from https://birdsna.org

Rodrigue, J. P., Slack, B., & Blank, S. (2017). Gateways and transport corridors in North America. In J. P. Rodrigue, C. Comotois, & B. Slack (Eds.), *The Geography of Transport Systems* (4th ed.). New York, USA: Routledge Taylor & Francis Group.

Rodrigues, A. S. L., Brooks, T. M., Butchart, S. H. M., Chanson, J., Cox, N., Hoffmann, M., & Stuart, S. N. (2014). Spatially explicit trends in the global conservation status of vertebrates. *PLoS One*, *9*(11), e113934.http://doi.org/10.1371/journal.pone.0113934

Rodrigues da Silva, Ú. S., & Matos, D. M. D. S. (2006). The invasion of *Pteridium aquilinum* and the impoverishment of the seed bank in fire prone areas of Brazilian Atlantic Forest. *Biodiversity and Conservation*, *15*(9), 3035–3043. http://doi.org/10.1007/s10531-005-4877-z

Rodríguez-Catón, M., Villalba, R., Morales, M., & Srur, A. (2016). Influence of droughts on *Nothofagus pumilio* forest decline across northern Patagonia, Argentina. *Ecosphere*, *7*(7), e01390. http://doi.org/10.1002/ecs2.1390

Rodríguez-Estrella, R., Donázar, J. A., & Hiraldo, F. (1998). Raptors as indicators of environmental change in the scrub habitat of Baja California Sur, Mexico. *Conservation Biology*, *12*(4), 921–925.

Rohde, K., Heap, M., & Heap, D. (1993). Rapoport's rule does not apply to marine teleosts and cannot explain latitudinal gradients in species richness. *The American Naturalist*, *142*(1), 1–16. http://doi.org/10.1086/285526

Rojas, R. T. (1991). La agricultura en la época prehispánica. In T. Rojas (Ed.), *La Agricultura en Tierras Mexicanas desde sus Orígenes hasta Nuestros Días* (pp. 15–138). Ciudad de México, México: Grijalbo.

Román-Cuesta, R. M., Salinas, N., Asbjornsen, H., Oliveras, I., Huaman, V., Gutiérrez, Y., Puelles, L., Kala, J., Yabar, D., Rojas, M., Astete, R., Jordán, D. Y., Silman, M., Mosandl, R., Weber, M., Stimm, B., Günter, S., Knoke, T., & Malhi, Y. (2011). Implications of fires on carbon budgets in Andean cloud montane forest: the importance of peat soils and tree resprouting. *Forest Ecology and Management*, *261*(11), 1987–1997. https://doi.org/10.1016/j.foreco.2011.02.025

Rooth, J. E., & Stevenson, J. C. (2000). Sediment deposition patterns in *Phragmites australis* communities: implications for coastal areas threatened by rising sea-level. *Wetlands Ecology and Management*, *8*(2), 173–183. http://doi.org/10.1023/a:1008444502859

Roque, J., & León, B. (2006). Endemic Orchidaceae of Peru. *Revista Peruana de Biologia*, *13*(2), 759S–878S. http://doi.org/10.15381/rpb.v13i2.1953

Rosauer, D. F., & Jetz, W. (2015). Phylogenetic endemism in terrestrial mammals. *Global Ecology and Biogeography*, *24*(2), 168–179. http://doi.org/10.1111/geb.12237

Rosauer, D., Laffan, S. W., Crisp, M. D., Donnellan, S. C., & Cook, L. G. (2009). Phylogenetic endemism: a new approach for identifying geographical concentrations of evolutionary history. *Molecular Ecology*, *18*(19), 4061–4072. http://doi.org/10.1111/j.1365-294X.2009.04311.x

Roy, K., Jablonski, D., Valentine, J. W., & Rosenberg, G. (1998). Marine latitudinal diversity gradients: tests of causal hypotheses. *PNAS*, *95*(7), 3699–3702. http://doi.org/10.1073/pnas.95.7.3699

Rozzi, R., Armesto, J. J., Goffinet, B., Buck, W., Massardo, F., Silander, J., Arroyo, M. T. K., Russell, S., Anderson, C. B., Cavieres, L. A., & Callicott, J. B. (2008). Changing lenses to assess biodiversity: patterns of species richness in sub-Antarctic plants and implications for global conservation. *Frontiers in Ecology and the Environment*, *6*(3), 131–137. https://doi.org/10.1890/070020

Ruiz, G. M., Fofonoff, P. W., Carlton, J. T., Wonham, M. J., & Hines, A. H. (2000). Invasion of coastal marine communities in North America: apparent patterns, processes, and biases. *Annual Review of Ecology and Systematics*, *31*(1), 481–531. http://doi.org/10.1146/annurev.ecolsys.31.1.481

Ruiz, G. M., Fofonoff, P. W., Steves, B. P., & Carlton, J. T. (2015). Invasion history and vector dynamics in coastal marine ecosystems: a North American perspective. *Aquatic Ecosystem Health & Management*, *18*(3), 299–311. http://doi.org/10.1080/14634988.2015.1027534

Rundel, P. W., Arroyo, M. T. K., Cowling, R. M., Keeley, J. E., Lamont, B. B., & Vargas, P. (2016). Mediterranean biomes: evolution of their vegetation, floras and climate. *Annual Review of Ecology, Evolution, and Systematics*, *47*, 383–407. http://doi.org/10.1146/annurev-ecolsys-121415-032330

Rundel, P. W., Dillon, M. O., Palma, B., Mooney, H. A., Gulmon, S. L., & Ehleringer, J. R. (1991). The phytogeography and ecology of the coastal Atacama and Peruvian deserts. *Aliso: A Journal of Systematic and Evolutionary Botany*, *13*(1), 1–49. http://doi.org/10.5642/aliso.19911301.02

Rutherford, S., D'Hondt, S., & Prell, W. (1999). Environmental controls on the geographic distribution of zooplankton diversity. *Nature*, *400*(6746), 749–753. http://doi.org/10.1038/23449

Ryberg, W. A., Hill, M. T., Painter, C. W., & Fitzgerald, L. A. (2014). Linking irreplaceable landforms in a self-organizing landscape to sensitivity of population vital rates for an ecological specialist. *Conservation Biology*, *29*(3), 888–898.

Rylander, K., Perez, J., & Gomez, J. (1996). Status of the green mussel, *Perna viridis* (Linnaeus, 1758)(Mollusca: Mytilidae), in northeastern Venezuela. *Caribbean Marine Studies*, *5*, 86–87.

Rzedowski, J. (1991). Diversidad y orígenes de la flora fanerogámica de México. *Acta Botánica Mexicana*, *14*, 3–21.

Saavedra, B., & Simonetti, J. A. (2005). Small mammals of Maulino forest remnants, a vanishing ecosystem of south-central Chile. *Mammalia*, *69*(3–4), 337–348. http://doi.org/10.1515/mamm.2005.027

Sacco, A. G., Bergmann, F. B., & Rui, A. M. (2013). Assembleia de aves na área urbana do município de Pelotas, Rio Grande do Sul, Brasil. *Biota Neotropica*, *13*(2), 153–162. http://doi.org/10.1590/S1676-06032013000200014

Sadovy, Y., & Eklund, A. (1999). *Synopsis of Biological Information on the Nassau Grouper, Epinephelus striatus (Bloch, 1792), and the Jewfish, E. itajara (Lichtenstein, 1822). NOAA Technical Report NMFS 146. Technical Report of the Fishery Bulletin. FAO Fisheries Synopsis 157.* Seattle, USA: US Department of Commerce. Retrieved from https://www.nrc.gov/docs/ML1224/ML12240A298.pdf

Sahin, V., & Hall, M. J. (1996). The effects of afforestation and deforestation on water yields. *Journal of Hydrology*, *178*(1), 293–309. http://doi.org/10.1016/0022-1694(95)02825-0

Sala, E., & Knowlton, N. (2006). Global marine biodiversity trends. *Annual Review of Environment and Resources*, *31*(1), 93–122. http://doi.org/10.1146/annurev.energy.31.020105.100235

Salazar, A., Baldi, G., Hirota, M., Syktus, J., & McAlpine, C. (2015). Land use and land cover change impacts on the regional climate of non-Amazonian South America: A review. *Global and Planetary Change*, *128*, 103–119. http://doi.org/10.1016/j.gloplacha.2015.02.009

Salvador, F., Monerris, J., & Rochefort, L. (2014). Peatlands of the Peruvian Puna ecoregion: types, characteristics and disturbance. *Mires and Peat*, *15*(4), 1–17.

Samson, F. and F. Knopf. 1994. Prairie Conservation in North America. BioScience 44:418-421.

Sandoval, M. L., & Barquez, R. M. (2013). The Chacoan bat fauna identity: Patterns of distributional congruence and conservation implications. *Revista Chilena de Historia Natural*, *86*(1), 75–94. http://doi.org/10.4067/S0716-078X2013000100007

Sanguinetti, A., & Singer, R. B. (2014). Invasive bees promote high reproductive success in Andean orchids. *Biological Conservation*, *175*, 10–20. http://doi.org/10.1016/j.biocon.2014.04.011

Santiago-Alarcon, D., Arriaga-Weiss, S. L., & Escobar, O. (2011). Bird community composition of Centla Marshes Biosphere Reserve, Tabasco, Mexico. *Ornitologia Neotropical*, *22*, 229–246.

Santos, R. O., Lirman, D., & Pittman, S. J. (2016). Long-term spatial dynamics in vegetated seascapes: fragmentation and habitat loss in a human-impacted subtropical lagoon. *Marine Ecology*, *37*(1), 200–214. http://doi.org/10.1111/maec.12259

Sanzo, D., & Hecnar, S. J. (2006). Effects of road de-icing salt (NaCl) on larval wood frogs (*Rana sylvatica*). *Environmental Pollution*, *140*(2), 247–256. http://doi.org/10.1016/j.envpol.2005.07.013

Sarukhán, J., Urquiza-Haas, T., Koleff, P., Carabias, J., Dirzo, R., Ezcurra, E., Cerdeira-Estrada, S., & Soberón., J. (2015). Strategic actions to value, conserve, and restore the natural capital of megadiversity countries: the case of Mexico. *BioScience*, *65*(2), 164–173. https://doi.org/10.1093/biosci/biu195

Scherber, C., Eisenhauer, N., Weisser, W. W., Schmid, B., Voigt, W., Fischer, M., Schulze, E.-D., Roscher, C., Weigelt, A., Allan, E., Bessler, H., Bonkowski, M., Buchmann, N., Buscot, F., Clement, L. W., Ebeling, A., Engels, C., Halle, S., Kertscher, I., Klein, A.-M., Koller, R., König, S., Kowalski, E., Kummer, V., Kuu, A., Lange, M., Lauterbach, D., Middelhoff, C., Migunova, V. D., Milcu, A., Muller, R., Partsch, S., Petermann, J. S., Renker, C., Rottstock, T., Sabais, A., Scheu, S., Schumacher, J., Temperton, V. M., & Tscharntke, T. (2010). Bottom-up effects of plant diversity on multitrophic interactions in a biodiversity experiment. *Nature*, *468*, 553–556. https://doi.org/10.1038/nature09492

Scherer-Lorenzen, M. (2014). The functional role of biodiversity in the context of global change. In D. A. Coomes, F. R. . Burslem, & W. D. Simonson (Eds.), *Forests and Global Change* (pp. 195–238). Cambridge, UK: Cambridge University Press.

Scherson, R. A., Thornhill, A. H., Urbina-Casanova, R., Freyman, W. A., Pliscoff, P. A., & Mishler, B. D. (2017). Spatial phylogenetics of the vascular flora of Chile. *Molecular Phylogenetics and Evolution*, *112*, 88–95. http://doi.org/10.1016/j.ympev.2017.04.021

Schmid-Hempel, R., Eckhardt, M., Goulson, D., Heinzmann, D., Lange, C., Plischuk, S., Escudero, L. R., Salathé, R., Scriven, J. J., & Schmid-Hempel, P. (2014). The invasion of southern South America by imported bumblebees and associated parasites. *Journal of Animal Ecology*, *83*(4), 823–837. https://doi.org/10.1111/1365-2656.12185

Schmidt, S. K., Nemergut, D. R., Miller, A. E., Freeman, K. R., King, A. J., & Seimon, A. (2009). Microbial activity and diversity during extreme freeze-thaw cycles in periglacial soils, 5400 m elevation, Cordillera Vilcanota, Perú. *Extremophiles: Microbial Life under Extreme Conditions*, *13*(5), 807–816. http://doi.org/10.1007/s00792-009-0268-9

Schulz, C., Koch, R., Cierjacks, A., & Kleinschmit, B. (2017). Land change and loss of landscape diversity at the Caatinga phytogeographical domain – Analysis of pattern-process relationships with MODIS land cover products (2001–2012). *Journal of Arid Environments*, *136*, 54–74. http://doi.org/10.1016/j.jaridenv.2016.10.004

Schulz, J. J., Cayuela, L., Echeverria, C., Salas, J., & Rey Benayas, J. M. (2010). Monitoring land cover change of the dryland forest landscape of Central Chile (1975-2008). *Applied Geography*, *30*(3), 436–447. http://doi.org/10.1016/j.apgeog.2009.12.003

Schulz, N., Aceituno, P., & Richter, M. (2011). Phytogeographic divisions, climate change and plant dieback along the coastal desert of northern Chile. *Erdkunde*, *65*(2), 169–187. http://doi.org/10.3112/erdkunde.2011.02.05

Schuur, E. A. G., McGuire, A. D., Schädel, C., Grosse, G., Harden, J. W., Hayes, D. J., Hugelius, G., Koven, C. D., Kuhry, P., Lawrence, D. M., Natali, S. M., Olefeldt, D., Romanovsky, V. E., Schaefer, K., Turetsky, M. R., Treat, C. C., & Vonk, J. E. (2015). Climate change and the permafrost carbon feedback. *Nature*, *520*(January 2016), 171–179. https://doi.org/10.1038/nature14338

Schuur, E. A. G., Vogel, J. G., Crummer, K. G., Lee, H., Sickman, J. O., & Osterkamp, T. E. (2009). The effect of permafrost thaw on old carbon release and net carbon exchange from tundra. *Nature*, *459*(7246), 556–559. http://doi.org/10.1038/nature08031

Schwartz, M. W., Thorne, J. H., & Viers, J. H. (2006). Biotic homogenization of the California flora in urban and urbanizing regions. *Biological Conservation*, *127*(3), 282–291. http://doi.org/10.1016/j.biocon.2005.05.017
Seebens, H., Blackburn, T. M., Dyer, E. E., Genovesi, P., Hulme, P. E., Jeschke, J. M., Pagad, S., Pyšek, P., Winter, M., Arianoutsou, M., Bacher, S., Blasius, B., Brundu, G., Capinha, C., Celesti-Grapow, L., Dawson, W., Dullinger, S., Fuentes, N., Jäger, H., Kartesz, J., Kenis, M., Kreft, H., Kühn, I., Lenzner, B., Liebhold, A., Mosena, A., Moser, D., Nishino, M., Pearman, D., Pergl, J., Rabitsch, W., Rojas-Sandoval, J., Roques, A., Rorke, S., Rossinelli, S., Roy, H. E., Scalera, R., Schindler, S., Štajerová, K., Tokarska-Guzik, B., van Kleunen, M., Walker, K., Weigelt, P., Yamanaka, T., & Essl, F. (2017). No saturation in the accumulation of alien species worldwide. *Nature Communications*, *8*, 14435. https://doi.org/10.1038/ncomms14435
Seebens, H., Schwartz, N., Schupp, P. J., & Blasius, B. (2016). Predicting the spread of marine species introduced by global shipping. *PNAS*, *113*(20), 5646–5651. http://doi.org/10.1073/pnas.1524427113
Serrano-Sandi, J., F. Bonilla-Murillo, & Sasa, M. (2013). Distribution, surface and protected area of palm-swamps in Costa Rica and Nicaragua. *Revista de Biologia Tropical*, *61*(Suppl. 1), 25–33.
Seto, K. C., Güneralp, B., & Hutyra, L. R. (2012). Global forecasts of urban expansion to 2030 and direct impacts on biodiversity and carbon pools. *PNAS*, *109*, 16083–16088. http://doi.org/10.1073/pnas.1211658109
Shaver, G. R., Laundre, J. A., Syndonia Bret-Harte, M., Stuart Chapin, F., Mercado-Díaz, J. A., Giblin, A. E., Gough, L., Gould, W. A., Hobbie, S. E., Kling, G. W., Mack, M. C., Moore, J. C., Nadelhoffer, K. J., Rastetter, E. B., & Schimel, J. P. (2014). Terrestrial ecosystems at Toolik Lake, Alaska. In J. E. Hobbie & G. W. Kling (Eds.), *Alaska's Changing Arctic: Ecological Consequences for Tundra, Streams, and Lakes*. New York, USA: Oxford University Press.
Shaver, I., Chain Guadarrama, A., Cleary, K., Sanfiorenzo, A. R., Santiago-Garcia, R., Finegan, B., Hormel, L., Sibelet, N., Vierling, L. A., Bosque-Perez, N., DeClerck, F., Fagan, M. E., & Waits, L. (2015). Coupled social and ecological outcomes of agricultural intensification in Costa Rica and the future of biodiversity conservation in tropical agricultural regions. *Global Environmental Change*, *32*, 74–86. https://doi.org/10.1016/j.gloenvcha.2015.02.006
Shochat, E., Lerman, S. B., Anderies, J. M., Warren, P. S., Faeth, S. H., & Nilon, C. H. (2010). Invasion, competition, and biodiversity loss in urban ecosystems. *BioScience*, *60*(3), 199–208. http://doi.org/10.1525/bio.2010.60.3.6
Sica, Y. V., Quintana, R. D., Radeloff, V. C., & G. I. Gavier-Pizarro. (2016). Wetland loss due to land use change in the Lower Paraná River Delta, Argentina. *Science of the Total Environment*, *568*, 967–978. http://doi.org/10.1016/j.scitotenv.2016.04.200
Siemann, E., Tilman, D., Haarstad, J., & Ritchie, M. (1998). Experimental tests of the dependence of arthropod diversity on plant diversity. *The American Naturalist*, *152*(2), 738–750. http://doi.org/10.1086/286204
Sierra, R. (2013). *Patrones y Factores de Deforestación en el Ecuador Continental, 1990-2010. Y un Acercamiento a los Próximos 10 Años*. Quito, Ecuador: Conservación Internacional Ecuador y Forest Trends.
Silva, J. M. C., Souza, M. A., Bieber, A. G. D., & Carlos, C. J. (2003). Aves da Caatinga: status, uso do habitat e sensitividade. In I. R. Leal, M. Tabarelli, & J. M. C. Silva (Eds.), *Ecologia e Conservação da Caatinga* (pp. 237–273). Racife, Brazil: Editora Universitaria, Universidade Federal de Pemambuco.
Silva, J. S. V., Abdon, M., Silva, S. M. A., & Moraes, J. A. (2011). Evolution of deforestation in the Brazilian pantanal and surroundings in the timeframe 1976-2008. *Geografia (Rio Claro)*, *36*(Especial), 35–55.
Simonetti, J. A. (1999). Diversity and conservation of terrestrial vertebrates in mediterranean Chile. *Revista Chilena de Historia Natural*, *72*(4), 493–500.
Simonetti, J. A., & Lazo, W. (1994). *Lepiota locaniensis*: an extinct Chilean fungus. *Revista Chilena De Historia Natural*, *67*(3), 351–352.
Singapore Index. (n.d.). Retrieved from http://www.cbd.int/authorities/gettinginvolved/cbi.shtml
Sklenář, P. (2016). Advance of plant species on slopes of the Chimborazo volcano (Ecuador) calculated based

on unreliable data. *PNAS*, *113*(4), E407–E408. http://doi.org/10.1073/pnas.1522531113

Sklenář, P., Dušková, E., & Balslev, H. (2011). Tropical and temperate: evolutionary history of páramo flora. *The Botanical Review*, *77*(2), 71–108. http://doi.org/10.1007/s12229-010-9061-9

Sklenář, P., Hedberg, I., & Cleef, A. M. (2014). Island biogeography of tropical alpine floras. *Journal of Biogeography*, *41*(2), 287–297. http://doi.org/10.1111/jbi.12212

Sklenář, P., Luteyn, J., Ulloa, C., Jorgensen, P., & Dillon, M. (2005). *Flora Genérica de los Páramos: Guía Ilustrada de las Plantas Vasculares* (Vol. 92). New York, USA: The New York Botanical Garden Press.

Slippers, B., Burgess, T., Pavlic, D., Ahumada, R., Maleme, H., Mohali, S., Rodas, C., & Wingfield, M. J. (2009). A diverse assemblage of Botryosphaeriaceae infect *Eucalyptus* in native and non-native environments. *Southern Forests: A Journal of Forest Science*, *71*(2), 101–110. https://doi.org/10.2989/SF.2009.71.2.3.818

Smith, B. D. (1994). The origins of agriculture in the Americas. *Evolutionary Anthropology: Issues, News, and Reviews*, *3*(5), 174–184. http://doi.org/10.1002/evan.1360030507

Smith, D. D. (1998). Iowa Prairie: Original extent and loss, preservation and recovery attempts. *The Journal of the Iowa Academy of Science*, *105*(3), 94–108.

Smith, K. F., Acevedo-Whitehouse, K., & Pedersen, A. B. (2009). The role of infectious diseases in biological conservation. *Animal Conservation*, *12*(1), 1–12. http://doi.org/10.1111/j.1469-1795.2008.00228.x

Smith, K. F., Behrens, M. D., Max, L. M., & Daszak, P. (2008). U.S. drowning in unidentified fishes: Scope, implications, and regulation of live fish import. *Conservation Letters*, *1*(2), 103–109. http://doi.org/10.1111/j.1755-263X.2008.00014.x

Smith, L. M. (1993). *Estimated Presettlement and Current Acres of Natural Plant Communities in Louisiana*. Baton Rouge, LA: Louisiana Natural Heritage Program, Louisiana Department of Wildlife and Fisheries.

Snapp, S. S., Mafongoya, P. L., & Waddington, S. (1998). Organic matter technologies for integrated nutrient management in smallholder cropping systems of southern Africa. *Agriculture, Ecosystems and Environment*, *71*(1–3), 185–200. http://doi.org/10.1016/S0167-8809(98)00140-6

Solar, R. R. de C., Barlow, J., Ferreira, J., Berenguer, E., Lees, A. C., Thomson, J. R., Louzada, J., Maués, M., Moura, N. G., Oliveira, V. H. F., Chaul, J. C. M., Schoereder, J. H., Vieira, I. C. G., Mac Nally, R., & Gardner, T. A. (2015). How pervasive is biotic homogenization in human-modified tropical forest landscapes? *Ecology Letters*, *18*(10), 1108–1118. https://doi.org/10.1111/ele.12494

Somarriba, E., Beer, J., Alegre-Orihuela, J., Andrade, H., Cerda, R., DeClerck, F., Detlefsen, G., Escalante, M., Giraldo, L., Ibrahim, M., Krishnamurthy, L., Mena-Mosquera, V., Mora- Degado, J., Orozco, L., Scheelje, M., & Campos, J. (2012). Mainstreaming agroforestry in Latin America. In P. K. R. Nair & D. Garrity (Eds.), *Agroforestry the Future of Global Land Use, Advances in Agroforestry* (pp. 429–453). Dordrecht, Netherlands: Springer.

Sorte, C. J. B., Williams, S. L., & Carlton, J. T. (2010). Marine range shifts and species introductions: comparative spread rates and community impacts. *Global Ecology and Biogeography*, *19*(3), 303–316. http://doi.org/10.1111/j.1466-8238.2009.00519.x

Soto-Azat, C., Peñafiel-Ricaurte, A., Price, S. J., Sallaberry-Pincheira, N., García, M. P., Alvarado-Rybak, M., & Cunningham, A. A. (2016). *Xenopus laevis* and emerging amphibian pathogens in Chile. *EcoHealth*, *13*(4), 775–783. http://doi.org/10.1007/s10393-016-1186-9

Soto-Azat, C., Valenzuela-Sánchez, A., Collen, B., Rowcliffe, J. M., Veloso, A., & Cunningham, A. A. (2013). The population decline and extinction of Darwin's frogs. *PLoS ONE*, *8*(6), e66957. http://doi.org/10.1371/journal.pone.0066957

Southwestern Center for Herpetological Research. (n.d.). Snakes of the American Southwest. Retrieved from http://www.southwesternherp.com/snakes

Souza, V., Eguiarte, L. E., Siefert, J., & Elser, J. J. (2008). Microbial endemism: does phosphorus limitation enhance speciation? *Nature Reviews Microbiology*, *6*(7), 559–564. http://doi.org/10.1038/nrmicro1917

Souza, V., Espinosa-Asuar, L., Escalante, A. E., Eguiarte, L. E., Farmer, J., Forney, L., Lloret, L., Rodríguez-Martínez, J. M., Soberón, X., Dirzo, R., & Elser, J. J. (2006). An endangered oasis of aquatic microbial biodiversity in the Chihuahuan desert. *PNAS*, *103*(17), 6565–6570.https://doi.org/10.1073/pnas.0601434103

Souza, V., Siefert, J. L., Escalante, A. E., Elser, J. J., & Eguiarte, L. E. (2012). The Cuatro Ciénegas Basin in Coahuila, Mexico: an astrobiological Precambrian park. *Astrobiology*, *12*(7), 641–647. http://doi.org/10.1089/ast.2011.0675

Spalding, M. D., Fish, L., & Wood, L. J. (2008). Toward representative protection of the world's coasts and oceans-progress, gaps, and opportunities. *Conservation Letters*, *1*(5), 217–226. http://doi.org/10.1111/j.1755-263X.2008.00030.x

Spera, S. A., Galford, G. L., Coe, M. T., Macedo, M. N., & Mustard, J. F. (2016). Land-use change affects water recycling in Brazil's last agricultural frontier. *Global Change Biology*, *22*(10), 3405–3413. http://doi.org/10.1111/gcb.13298

Spracklen, D. V., & Righelato, R. (2014). Tropical montane forests are a larger than expected global carbon store. *Biogeosciences*, *11*(10), 2741–2754. http://doi.org/10.5194/bg-11-2741-2014

Spriggs, E. L., Christinb, P. A., & Edwards, E. J. (2014). C4 photosynthesis promoted species diversification during the miocene grassland expansion. *PLoS ONE*, *9*(5), e97722. http://doi.org/10.1371/journal.pone.0097722

Sproull, G. J., Quigley, M. F., Sher, A., & González, E. (2015). Long-term changes in composition, diversity and distribution patterns in four herbaceous plant communities along an elevational gradient. *Journal of Vegetation Science*, *26*(3), 552–563. http://doi.org/10.1111/jvs.12264

Squeo, F. A., Cavieres, L. A., Arancio, G., Novoa, J. E., Matthei, O., Marticorena, C., Rodriguez, R., Arroyo, M. T. K., & Munoz, M. (1998). Biodiversity of vascular flora in the Antofagasta Region, Chile. *Revista Chilena de Historia Natural*, *71*(4), 571–591.

Stapp, P., Antolin, M. F., & Ball, M. (2004). Patterns of extinction in prairie dog metapopulations: plague outbreaks follow El Niño events. *Frontiers in Ecology and the Environment*, *2*(5), 235–240. http://doi.org/10.1890/1540-9295(2004)002%5B0235:POEIPD%5D2.0.CO;2

Stattersfield, A. J., Crosby, M. J., Long, A. J., & Wege, D. C. (1998). *Endemic Bird Areas of the World: Priorities for Biodiversity Conservation*. Cambridge, UK: BirdLife International.

Steele, M. K., Heffernan, J. B., Bettez, N., Cavender-Bares, J., Groffman, P. M., Grove, J. M., Hall, S., Hobbie, S. E., Larson, K., Morse, J. L., Neill, C., Nelson, K. C., O'Neil-Dunne, J., Ogden, L., Pataki, D. E., Polsky, C., & Roy Chowdhury, R. (2014). Convergent surface water distributions in U.S. cities. *Ecosystems*, *17*(4), 685–697. https://doi.org/10.1007/s10021-014-9751-y

Stein, B. A., Kutner, L. S., & Adams, J. S. (Eds.). (2000). *Precious Heritage: The Status of Biodiversity in the United States*. New York, USA: Oxford University Press.

Stephenson, N. L., & Das, A. J. (2011). Comment on "Changes in climatic water balance drive downhill shifts in plant species' optimum elevations." *Science*, *334*(6053), 177c. http://doi.org/10.1126/science.1205740

Stevens, N., Lehmann, C. E. R., Murphy, B. P., & Durigan, G. (2017). Savanna woody encroachment is widespread across three continents. *Global Change Biology*, *23*(1), 235–244. http://doi.org/10.1111/gcb.13409

Stewart, J. A., Perrine, J. D., Nichols, L. B., Thorne, J. H., Millar, C. I., Goehring, K. E., & Wright, D. H. (2015). Revisiting the past to foretell the future: summer temperature and habitat area predict pika extirpations in California. *Journal of Biogeography*, *42*(5), 880–890. http://doi.org/10.1111/jbi.12466

Stöhr, S., O'Hara, T. D., & Thuy, B. (2012). Global diversity of brittle stars (Echinodermata: Ophiuroidea). *PLoS ONE*, *7*(3), e31940. http://doi.org/10.1371/journal.pone.0031940

Stotz, D. F., Fitzpatrick, J. W., Parker III, T., & Moskovits, D. K. (1996). *Neotropical Birds: Ecology and Conservation*. Chicago, USA: The University of Chicago Press.

Stranko, S. A., Gresens, S. E., Klauda, R. J., Kilian, J. V., Ciccotto, P. J., Ashton, M. J., & Becker, A. J. (2010). Differential effects of urbanization and non-natives on imperiled stream species. *Northeastern Naturalist*, *17*(4), 593–614. http://doi.org/10.1656/045.017.0406

Stuart, S. N., Hoffmann, M., Chanson, J. S., Cox, N. A., Berridge, R. J., Ramani, P., & Young, B. E. (Eds.). (2008). *Threatened Amphibians of the World*. Barcelona, Spain; Gland Switzerland; Arlington, USA: Lynx Edicions, IUCN and Conservation International.

Sturm, M., Racine, C., & Tape, K. (2001). Climate change: increasing shrub abundance in the Arctic. *Nature*, *411*(6837), 546–547. http://doi.org/10.1038/35079180

Suarez, M. L., Ghermandi, L., & Kitzberger, T. (2004). Factors predisposing episodic drought-induced tree mortality in *Nothofagus* - site, climatic sensitivity and growth trends. *Journal of Ecology*, *92*(6), 954–966. http://doi.org/10.1111/j.1365-2745.2004.00941.x

Sutherland, K. P., Porter, J. W., Turner, J. W., Thomas, B. J., Looney, E. E., Luna, T. P., Meyers, M. K., Futch, J. C., & Lipp, E. K. (2010). Human sewage identified as likely source of white pox disease of the threatened Caribbean elkhorn coral, *Acropora palmata*. *Environmental Microbiology*, *12*(5), 1122–1131. https://doi.org/10.1111/j.1462-2920.2010.02152.x

Sutherland, R. W., Dunning, P. R., & Baker, W. M. (2010). Amphibian encounter rates on roads with different amounts of traffic and urbanization. *Conservation Biology*, *24*(6), 1626–1635. http://doi.org/10.1111/j.1523-1739.2010.01570.x

Suzán, G., Marcé, E., Giermakowski, J. T., Mills, J. N., Ceballos, G., Ostfeld, R. S., Armién, B., Pascale, J. M., & Yates, T. L. (2009). Experimental evidence for reduced rodent diversity causing increased hantavirus prevalence. *PLoS ONE*, *4*(5), e5461. https://doi.org/10.1371/journal.pone.0005461

Svriz, M., Damascos, M. A., Zimmermann, H., & Hensen, I. (2013). The exotic shrub *Rosa rubiginosa* as a nurse plant. Implications for the restoration of disturbed temperate forests in Patagonia, Argentina. *Forest Ecology and Management*, *289*, 234–242. http://doi.org/10.1016/j.foreco.2012.09.037

Swenson, N. G., Enquist, B. J., Pither, J., Kerkhoff, A. J., Boyle, B., Weiser, M. D., Elser, J. J., Fagan, W. F., Forero-Montana, J., Fyllas, N., Kraft, N. J. B., Lake, J. K., Moles, A. T., Patino, S., Phillips, O. L., Price, C. A., Reich, P. B., Quesada, C. A., Stegen, J. C., Valencia, R., Wright, I. J., Wright, S. J., Andelman, S., Jorgensen, P. M., Lacher, T. E., Monteagudo, A., Nunez-Vargas, M. P., Vasquez-Martinez, R., & Nolting, K. M. (2012). The biogeography and filtering of woody plant functional diversity in North and South America. *Global Ecology and Biogeography*, *21*(8), 798–808. https://doi.org/10.1111/j.1466-8238.2011.00727.x

Syphard, A. D., Keeley, J. E., & Abatzoglou, J. T. (2017). Trends and drivers of fire activity vary across California aridland ecosystems. *Journal of Arid Environments*, *144*, 110–122. http://doi.org/10.1016/j.jaridenv.2017.03.017

Tabarelli, M., Aguiar, A. V., Ribeiro, M. C., Metzger, J. P., & Peres, C. A. (2010). Prospects for biodiversity conservation in the Atlantic Forest: lessons from aging human-modified landscapes. *Biological Conservation*, *143*(10), 2328–2340. http://doi.org/10.1016/j.biocon.2010.02.005

Tacón, A., Palma, J., Fernández, U., & Ortega, F. (2006). *El Mercado de los Productoss Forestales no Madereros y la Conservation de los Bosques del Sur de Chile y Argentina*. Valdivia, Chile: WWF Chile and Red de Productos Forestales No Madereros de Chile.

Tape, K., Sturm, M., & Racine, C. (2006). The evidence for shrub expansion in Northern Alaska and the Pan-Arctic. *Global Change Biology*, *12*(4), 686–702. http://doi.org/10.1111/j.1365-2486.2006.01128.x

Tavares, R., & Lopez, D. (2009). Fishery production trends of elasmobranchs from Venezuela: with emphasis on sharks. *Proceedings of the 62nd Gulf and Caribbean Fisheries Institute*, *62*, 178–183.

Taylor, B. L., Rojas-Bracho, L., Moore, J., Jaramillo-Legorreta, A., Ver Hoef, J. M., Cardenas-Hinojosa, G., Nieto-Garcia, E., Barlow, J., Gerrodette, T., Tregenza, N., Thomas, L., & Hammond, P. S. (2017). Extinction is imminent for Mexico's endemic porpoise unless fishery bycatch is eliminated. *Conservation Letters*, *10*(5), 588–595. https://doi.org/10.1111/conl.12331

Taylor, K. T., Maxwell, B. D., Pauchard, A., Nuñez, M. A., & Rew., L. J. (2016). Native versus non-native invasions: similarities and differences in the biodiversity impacts of *Pinus contorta* in introduced and native ranges. *Diversity and Distributions*, *22*(5), 578–588. http://doi.org/10.1111/ddi.12419

Tecco, P. A., Gurvich, D. E., Díaz, S., Pérez-Harguindeguy, N., & Cabido, M. (2006). Positive interaction between invasive plants: the influence of Pyracantha angustifolia on the recruitment of native and exotic woody species. *Austral Ecology*, *31*(3), 293–300. http://doi.org/10.1111/j.1442-9993.2006.01557.x

Tedersoo, L., Bahram, M., Põlme, S., Kõljalg, U., Yorou, N. S., Wijesundera, R., Ruiz, L. V., Vasco-Palacios, A. M., Thu, P. Q., Suija, A., Smith, M. E., Sharp, C., Saluveer, E., Saitta, A., Rosas, M., Riit, T., Ratkowsky, D., Pritsch, K., Poldmaa, K., Piepenbring, M., Phosri, C., Peterson, M., Parts, K., Partel, K., Otsing, E., Nouhra, E., Njouonkou, A. L., Nilsson, R. H., Morgado, L. N., Mayor, J., May, T. W., Majuakim, L., Lodge, D. J., Lee, S. S., Larsson, K.-H., Kohout, P., Hosaka, K., Hiiesalu, I., Henkel, T. W., Harend, H., Guo, L. D., Greslebin, A., Grelet, G., Geml, J., Gates, G., Dunstan, W., Dunk, C., Drenkhan, R., Dearnaley, J., De Kesel, A., Dang, T., Chen, X., Buegger, F., Brearley, F. Q., Bonito, G., Anslan, S., Abell, S., & Abarenkov, K. (2014). Global diversity and geography of soil fungi. *Science*, *346*(6213), 1256688.https://doi.org/10.1126/science.1256688

Teixeira, A. M. G., Soares-Filho, B. S., Freitas, S. R., & Metzger, J. P. (2009). Modeling landscape dynamics in an Atlantic Rainforest region: implications for conservation. *Forest Ecology and Management*, *257*(4), 1219–1230. http://doi.org/10.1016/j.foreco.2008.10.011

Tejedor-Garavito, N., Álvarez Dávila, E., Arango Caro, S., Araujo Murakami, A., Blundo, C., Boza Espinoza, T. E., La Torre Cuadros, M. A., Gaviria, J., Gutiérrez, N., Jørgensen, P. M., León, B., López Camacho, R., Malizia, L., Timaná de la Flor, M., Ulloa Ulloa, C., Vacas Cruz, O., & Newton, A. C. (2012). Evaluación del estado de conservación de los bosques montanos en los Andes tropicales. *Ecosistemas*, *21*(1–2), 148–166.

Tejedor Garavito, N., Álvarez Dávila, E., Arango Caro, S., Araujo Murakami, A., Baldeón, A., Beltrán, H., Blundo, C., Boza Espinoza, T. E., Fuentes Claros, A., Gaviria, J., Gutiérrez, N., Khela, S., León, B., La Torre Cuadros, M. A., López Camacho, R., Malizia, L., Millán, B., Moraes, R. M., Newton, A., Pacheco, S., Reynel, C., Ulloa Ulloa, C., & Vacas Cruz, O. (2014). *A Regional Red List of Montane Tree Species of the Tropical Andes: Trees at the top of the world*. Richmond, UK: Botanic Gardens Conservation International.

Tejedor Garavito, N., Newton, A. C., Golicher, D., & Oldfield, S. (2015). The relative impact of climate change on the extinction risk of tree species in the montane tropical Andes. *PLoS ONE*, *10*(7), e0131388. http://doi.org/10.1371/journal.pone.0131388

Tellería, J. L., Venero, J. L., & Santos, T. (2006). Conserving birdlife of Peruvian highland bogs: effects of patch-size and habitat quality on species richness and bird numbers. *Ardeola*, *53*(2), 271–283.

Temperton, V. M., Mwangi, P. N., Scherer-Lorenzen, M., Schmid, B., & Buchmann, N. (2007). Positive interactions between nitrogen-fixing legumes and four different neighbouring species in a biodiversity experiment. *Oecologia*, *151*(2), 190–205. http://doi.org/10.1007/s00442-006-0576-z

ter Steege, H., Pitman, N. C. A., Killeen, T. J., Laurance, W. F., Peres, C. A., Guevara, J. E., Salomão, R. P., Castilho, C. V., Amaral, I. L., de Almeida Matos, F. D., Valderrama Sandoval, E. H., & Valenzuela Gamarra, L. (2015). Estimating the global conservation status of more than 15,000 Amazonian tree species. *Science Advances*, *1*(10), e1500936. https://doi.org/10.1126/sciadv.1500936

ter Steege, H., Pitman, N. C. A., Sabatier, D., Baraloto, C., Salomão, R. P., Guevara, J. E., Phillips, O. L., Castilho, C. V., Magnusson, W. E., Molino, J. F., Monteagudo, A., Nunez Vargas, P., Montero, J. C., Feldpausch, T. R., Coronado, E. N. H., Killeen, T. J., Mostacedo, B., Vasquez, R., Assis, R. L., Terborgh, J., Wittmann, F., Andrade, A., Laurance, W. F., Laurance, S. G. W., Marimon, B. S., Marimon, B. H., Guimaraes Vieira, I. C., Amaral, I. L., Brienen, R., Castellanos, H., Cardenas Lopez, D., Duivenvoorden, J. F., Mogollon, H. F., Matos, F. D. D. A., Davila, N., Garcia-Villacorta, R., Stevenson Diaz, P. R., Costa, F., Emilio, T., Levis, C., Schietti, J., Souza, P., Alonso, A., Dallmeier, F., Montoya, A. J. D., Fernandez Piedade, M. T., Araujo-Murakami, A., Arroyo, L., Gribel, R., Fine, P. V. A., Peres, C. A., Toledo, M., Aymard C., G. A., Baker, T. R., Ceron, C., Engel, J., Henkel, T. W., Maas, P., Petronelli, P., Stropp, J., Zartman, C. E., Daly, D., Neill, D., Silveira, M., Paredes, M. R., Chave, J., Lima Filho, D. D. A., Jorgensen, P. M., Fuentes, A., Schongart, J., Cornejo Valverde, F., Di Fiore, A., Jimenez, E. M., Penuela Mora, M. C., Phillips, J. F., Rivas, G., van Andel, T. R., von Hildebrand, P., Hoffman, B., Zent, E. L., Malhi, Y., Prieto, A., Rudas, A., Ruschell, A. R., Silva, N., Vos, V. A., Zent, S., Oliveira, A. A., Schutz, A. C., Gonzales, T., Trindade Nascimento, M., Ramirez-Angulo, H., Sierra, R., Tirado, M., Umana Medina, M. N., van der Heijden, G., Vela, C. I. A., Vilanova Torre, E., Vriesendorp, C., Wang, O., Young, K. R., Baider, C., Balslev, H., Ferreira, C., Mesones, I., Torres-Lezama, A., Urrego Giraldo, L. E., Zagt, R., Alexiades, M. N., Hernandez, L., Huamantupa-Chuquimaco, I., Milliken, W., Palacios Cuenca, W., Pauletto, D., Valderrama Sandoval, E., Valenzuela Gamarra, L., Dexter, K. G., Feeley, K., Lopez-Gonzalez, G., & Silman, M. R. (2013). Hyperdominance in the Amazonian tree flora. *Science*, *342*(6156), 1243092. https://doi.org/10.1126/science.1243092 ter Steege, H., Vaessen, R. W., Cárdenas-López, D., Sabatier, D., Antonelli, A., de Oliveira, S. M., Pitman, N. C. A., Jørgensen, P. M., & Salomão, R. P. (2016). The discovery of the Amazonian tree flora with an updated checklist of all known tree taxa. *Scientific Reports*, *6*(1), 29549. https://doi.org/10.1038/srep29549

The Nature Conservancy. (2005). *Assessing Linkages between Agriculture and Biodiversity in Central America: Historical Overview and Future Perspectives*. San José, Costa Rica: Mesoamerican & Caribbean Region, Conservation Science Program.

The Nature Conservancy. (2009). TNC Terrestrial Ecoregions. Retrieved May 1, 2016, from http://maps.tnc.org/gis_data.html

Thébault, E., & Loreau, M. (2003). Food-web constraints on biodiversity-ecosystem functioning relationships. *PNAS*, *100*(25), 14949–14954. http://doi.org/10.1073/pnas.2434847100

Thomas, E., Vandebroek, I., Goetghebeur, P., Sanca, C., Arrázola, S., & Van Damme, P. (2008). The relationship between plant use and plant diversity in the Bolivian Andes, with special reference to medicinal plant use. *Human Ecology*, *36*(6), 861–879. http://doi.org/10.1007/s10745-008-9208-z

Thompson, D. Q., Stuckey, R. L., & Thompson, E. B. (1987). *Spread, Impact, and Control of Purple Loosestrife (Lythrum salicaria) in North American Wetlands*. Washington, D.C.: U.S. Fish and Wildlife Service.

Thorne, T., & Williams, E. S. (1988). Disease and endangered species: the black-footed ferret as a recent example. *Conservation Biology*, *2*(1), 66–74. http://doi.org/10.1111/j.1523-1739.1988.tb00336.x

Thornton, P. K. (2010). Livestock production: recent trends, future prospects. *Philosophical Transactions of the Royal Society of London B: Biological Sciences*, *365*(1554), 2853–2867. http://doi.org/10.1098/rstb.2010.0134

Tilman, D., Lehman, C. L., & Thomson, K. T. (1997). Plant diversity and ecosystem productivity: theoretical considerations. *PNAS*, *94*(5), 1857–1861.

Tilman, D., Reich, P. B., & Isbell, F. (2012). Biodiversity impacts ecosystem productivity as much as resources, disturbance, or herbivory. *PNAS*, *109*(26), 10394–10397. http://doi.org/10.1073/pnas.1208240109

Tingley, M. W., Monahan, W. B., Beissinger, S. R., & Moritz, C. (2009). Birds track their Grinnellian niche through a century of climate change. *PNAS*, *106*(Supplement 2), 19637–19643.http://doi.org/10.1073/pnas.0901562106

Tirado, C., Cortés, A., Carretero, M. A., & Bozinovic, F. (2016). Does the presence of livestock alter the trophic behaviour of sympatric populations of wild camelids *Vicugna vicugna* Molina 1782 and *Lama guanicoe* Müller 1976 (Artiodactyla: Camelidae)? Evidence from central Andes. *Gayana*, *80*(1), 29–39. http://doi.org/10.4067/S0717-65382016000100004

Tittensor, D. P., Mora, C., Jetz, W., Lotze, H. K., Ricard, D., Vanden Berghe, E., & Worm, B. (2010). Global patterns and predictors of marine biodiversity across taxa. *Nature*, *466*(7310), 1098–1101. http://doi.org/10.1038/nature09329

Tomasella, J., Neill, C., Figueiredo, R., & Nobre, A. D. (2009). Water and chemical budgets at the catchment scale including nutrient exports from intact forests and disturbed landscapes. In M. Keller, M. Bustamante, J. Gash, & P. S. Dias (Eds.), *Amazonia and Global Change* (pp. 505–524). Washington D.C., USA: American Geophysical Union.

Toral-Granda, M. V., Causton, C. E., Jäger, H., Trueman, M., Izurieta, J. C., Araujo, E., Cruz, M., Zander, K. K., Izurieta, A., & Garnett, S. T. (2017). Alien species pathways to the Galapagos Islands, Ecuador. *PLoS ONE*, *12*(9), e0184379. https://doi.org/10.1371/journal.pone.0184379

Tordoff, H. ., & Redig, P. T. (2001). Role of genetic background in the success of reintroduced peregrine falcons. *Conservation Biology*, *15*(2), 528–532. http://doi.org/10.1046/j.1523-1739.2001.015002528.x

Torres-Guevara, J., Parra-Rodinel, F., & Casas, A. (2017). Panorama de los recursos genéticos en Perú. In A. Casas, J. Torres-Guevara, & F. Parra-Rondinel (Eds.), *Domesticación en el Continente Americano Vol. 2. Investigación para el Manejo Sustentable de Recursos Genéticos en el Nuevo Mundo* (pp. 103–133). México D.F., México and Lima, Perú: Universidad Nacional Autónoma de México/Universidad Nacional Agraria La Molina/CONACYT.

Tovar, C., Seijmonsbergen, A. C., & Duivenvoorden, J. F. (2013). Monitoring land use and land cover change in mountain regions: An example in the Jalca grasslands of the Peruvian Andes. *Landscape and*

Urban Planning, *112*, 40–49. http://doi.org/10.1016/J.LANDURBPLAN.2012.12.003

Trabucco, A., Zomer, R., Bossio, D., van Straaten, O., & Verchot, L. (2008). Climate change mitigation through afforestation/reforestation: a global analysis of hydrologic impacts with four case studies. *Agriculture, Ecosystems & Environment*, *126*(1–2), 81–97. http://doi.org/10.1016/j.agee.2008.01.015

Tracewski, Ł., Butchart, S. H. M., Di Marco, M., Ficetola, G. F., Rondinini, C., Symes, A., Wheatley, H., Beresford, A. E., & Buchanan, G. M. (2016). Toward quantification of the impact of 21st-century deforestation on the extinction risk of terrestrial vertebrates. *Conservation Biology*, *30*(5), 1070–1079. https://doi.org/10.1111/cobi.12715

Tree flora of the Neotropical Region. (n.d.). Retrieved from http://prof.icb.ufmg.br/treeatlan/treeatlanE_0_index.htm

Tremblay, B., Lévesque, E., & Boudreau, S. (2012). Recent expansion of erect shrubs in the Low Arctic: evidence from Eastern Nunavik. *Environmental Research Letters*, *7*(3), 35501. http://doi.org/10.1088/1748-9326/7/3/035501

Trexler, J. C., & Loftus, W. F. (2016). Invertebrates of the Florida Everglades. In D. Batzer & D. Boix (Eds.), *Invertebrates in Freshwater Wetlands* (pp. 321–356). Cham, Switzerlan: Springer.

Tribuzio, C. A., Rodgveller, C., Echave, K. B., & Hulson, P. J. (2015). Assessment of the shark stock complex in the Gulf of Alaska. In *Stock Assessment and Fishery Evaluation Report for the Groundfish Resources of the Gulf of Alaska for 2011* (pp. 1569–1642). Anchorage, USA.

Tuanmu, M. N., & Jetz, W. (2014). A global 1-km consensus land-cover product for biodiversity and ecosystem modellingxxx. *Global Ecology and Biogeography*, *23*(9), 1031–1045. http://doi.org/10.1111/geb.12182

Tubelis, D. P., & Cavalcanti, R. B. (2000). A comparison of bird communities in natural and disturbed non-wetland open habitats in the Cerrado's central region, Brazil. *Bird Conservation International*, *10*(4), 331–350.

Turnbull, L. A., Isbell, F., Purves, D. W., Loreau, M., & Hector, A. (2016). Understanding the value of plant diversity for ecosystem functioning through niche theory. *Proceedings of the Royal Society B: Biological Sciences*, *283*(1844), 20160536. https://doi.org/10.1098/rspb.2016.0536

Turner, R. E., Howes, B. L., Teal, J. M., Milan, C. S., Swenson, E. M., & Goehringer-Tonerb, D. D. (2009). Salt marshes and eutrophication: an unsustainable outcome. *Limnology and Oceanography*, *54*(5), 1634–1642. http://doi.org/10.4319/lo.2009.54.5.1634

U.S. Forestry Service. (2015). *National Report on Sustainable Forests*. United States Department of Agriculture.

Ulloa Ulloa, C., Acevedo-Rodríguez, P., Beck, S., Belgrano, M. J., Bernal, R., Berry, P. E., Brako, L., Celis, M., Davidse, G., Forzza, R. C., Gradstein, S. R., Hokche, O., León, B., León-Yánez, S., Magill, 1 R.E., Neill, D. A., Nee, M., Raven, P. H., Stimmel, S., Strong, M. T., Villaseñor, H. L., Zarucchi, J. L., Zuloaga, F. O., & Jørgensen, P. M. (2017). An integrated assessment of the vascular plants of the Americas. *Science*, *358*, 1614–1617.https://doi.org/10.1126/science.aao0398

UNEP-WCMC; IUCN. (2017). Protected Planet. Retrieved from https://www.protectedplanet.net/

UNESCO. (2002). Declaración Universal Sobre la Diversidad Cultural: Una Visión, una Plataforma Conceptual, un Semillero de Ideas, un Paradigma Nuevo. Johannesburgo, South Africa: Serie sobre la Diversidad Cultural. Documento preparado para la Cumbre Mundial sobre el Desarrollo Sostenible.

United Nations Development Programme. (2014). *Human Development Report 2014. Sustaining Human Progress: Reducing Vulnerabilities and Building Resilience*. New York, USA: United Nations Development Programme (UNDP).

Upham, N. S. (2017). Past and present of insular Caribbean mammals: understanding Holocene extinctions to inform modern biodiversity conservation. *Journal of Mammalogy*, *98*(4), 913–917. http://doi.org/10.1093/jmammal/gyx079

Urban, M. C., Skelly, D. K., Burchsted, D., Price, W., & Lowry, S. (2006). Stream communities across a rural-urban landscape gradient. *Diversity and Distributions*, *12*(4), 337–350. http://doi.org/10.1111/j.1366-9516.2005.00226.x

Urbina, J. C., & Benavides, J. C. (2015). Simulated small scale disturbances increase decomposition rates and facilitates invasive species encroachment in a high elevation tropical Andean peatland. *Biotropica*, *47*(2), 143–151. http://doi.org/10.1111/btp.12191

USA quickfacts census. (2012). Retrieved June 16, 2013, from https://web.archive.org/web/20120304192040/http://quickfacts.census.gov:80/qfd/states/00000.html

US Fish and Wildlife Service. (2011). Submerged Aquatic Vegetation: Where Have All the Grasses Gone? Retrieved from http://www.fws.gov/chesapeakebay/cbsav.html

US Forest Service. (2016). *New Aerial Survey Identifies More Than 100 Million Dead Trees in California*. Retrieved from https://www.fs.fed.us/news/releases/new-aerial-survey-identifies-more-100-million-dead-trees-california

USDA. (n.d.). Plant Database. Retrieved from http://plants.usda.gov

USDA. (2007). *Land Use Status and Trends 2007*. Retrieved from https://www.nrcs.usda.gov/wps/portal/nrcs/detail/national/home/?cid=stelprdb1083124

USDA. (2015). *Major Forest Insect and Disease Conditions in the United States: 2013*. Washington D.C., USA: US Forest service.

USDA, & USDOI. (2012). LANDFIRE: Vegetation Departure 1.3.0. Retrieved from http://www.landfire.gov

USGS. (2003). *PRAIRIEMAP, A GIS Database for Prairie Grassland Management in Western North America*. Washington D.C., USA: USGS Forest and Rangeland Ecosystem, Science Center, Snake River Field Station.

Valdivia, C. E., Simonetti, J. A., & Henríquez, C. A. (2006). Depressed pollination of *Lapageria rosea* Ruiz et Pav. (Philesiaceae) in the fragmented temperate rainforest of southern South America. *Biodiversity and Conservation*, *15*(5), 1845–1856. http://doi.org/10.1007/s10531-004-6683-4

Valiela, I. (2006). *Global Coastal Change*. Malden, USA: Blackwell Publishing.

Valiela, I., Bowen, J. L., & York, J. K. (2001). Mangrove forests: one of the world's threatened major tropical environments. *BioScience*, *51*(10), 807. http://doi.org/10.1641/0006-3568(2001)051%5B0807:MFOOTW%5D2.0.CO;2

Van Kleunen, M., Dawson, W., Essl, F., Pergl, J., Winter, M., Weber, E., Kreft, H., Weigelt, P., Kartesz, J., Nishino, M., Antonova, L. A., Barcelona, J. F., Cabezas, F. J., Cárdenas, D., Cárdenas-Toro, J., N, C., Chacón, E., Chatelain, C., Ebel, A. L., Figueiredo, E., Fuentes, N., Groom, Q. J., Henderson, L., Inderjit, L., Kupriyanov, A., S, M., Morozova, O., Moser, D., Nickrent, D. L., Patzelt, A., Pelser, P. B., Baptiste, M. P., Poopath, M., Schulze, M., Seebens, H., Shu, W.-S., Thomas, J., Velayos, M., Wieringa, J. J., & Pyšek, P. (2015). Global exchange and accumulation of non-native plants. *Nature*, *525*(7567), 100–103. https://doi.org/10.1038/nature14910

van Ruijven, J., & Berendse, F. (2005). Diversity-productivity relationships: initial effects, long-term patterns, and underlying mechanisms. *PNAS*, *102*(3), 695–700. http://doi.org/10.1073/pnas.0407524102

van Ruijven, J., & Berendse, F. (2009). Long-term persistence of a positive plant diversity-productivity relationship in the absence of legumes. *Oikos*, *118*(1), 101–106. http://doi.org/10.1111/j.1600-0706.2008.17119.x

van Tussenbroek, B. I., Cortés, J., Collin, R., Fonseca, A. C., Gayle, P. M. H., Guzmán, H. M., Jácome, G. E., Juman, R., Koltes, K. H., Oxenford, H. A., Rodríguez-Ramirez, A., Samper-Villarreal, J., Smith, S. R., Tschirky, J. J., & Weil, E. (2014). Caribbean-wide, long-term study of seagrass beds reveals local variations, shifts in community structure and occasional collapse. *PLoS ONE*, *9*(3), e90600. https://doi.org/10.1371/journal.pone.0090600

Van Vliet, N., Mertz, O., Heinimann, A., Langanke, T., Pascual, U., Schmook, B., Adams, C., Schmidt-Vogt, D., Messerli, P., Leisz, S., Castella, J.-C., Jørgensen, L., Birch-Thomsen, T., Hett, C., Bech-Bruun, T., Ickowitz, A., Vu, K. C., Yasuyuki, K., Fox, J., Padoch, C., Dressler, W., & Ziegler, A. D. (2012). Trends, drivers and impacts of changes in swidden cultivation in tropical forest-agriculture frontiers: a global assessment. *Global Environmental Change*, *22*(2), 418–429. https://doi.org/10.1016/j.gloenvcha.2011.10.009

Vanderbilt, K., & Gaiser, E. (2017). The International Long Term Ecological Research Network: a platform for collaboration. *Ecosphere*, *8*(2), e01697. http://doi.org/10.1002/ecs2.1697

Vandermeer, J. (1981). The interference production principle: an ecological theory for agriculture.

BioScience, *31*(5), 361–364.http://doi.org/10.2307/1308400
Vásquez, D. L. A., Balslev, H., & Sklenář, P. (2015). Human impact on tropical-alpine plant diversity in the northern Andes. *Biodiversity and Conservation*, *24*(11), 2673–2683. http://doi.org/10.1007/s10531-015-0954-0
Vassallo, M. M., Dieguez, H. D., Garbulsky, M. F., Jobbágy, E. G., & Paruelo, J. M. (2013). Grassland afforestation impact on primary productivity: a remote sensing approach. *Applied Vegetation Science*, *16*(3), 390–403. http://doi.org/10.1111/avsc.12016
Vázquez, D. P. (2002). Multiple effects of introduced mammalian herbivores in a temperate forest. *Biological Invasions*, *4*(1–2), 175–191. http://doi.org/10.1023/A:1020522923905
Vega Thurber, R. L., Burkepile, D. E., Fuchs, C., Shantz, A. A., Mcminds, R., & Zaneveld, J. R. (2014). Chronic nutrient enrichment increases prevalence and severity of coral disease and bleaching. *Global Change Biology*, *20*(2), 544–554. http://doi.org/10.1111/gcb.12450
Veldman, J. W., Overbeck, G. E., Negreiros, D., Mahy, G., Le Stradic, S., Fernandes, G. W., Durigan, G., Buisson, E., Putz, F. E., & Bond, W. J. (2015). Where tree planting and forest expansion are bad for biodiversity and ecosystem services. *BioScience*, *65*(10), 1011–1018. https://doi.org/10.1093/biosci/biv118
Vences, M., & Köhler, J. (2008). Global diversity of amphibians (Amphibia) in freshwater. *Hydrobiologia*, *595*(1), 569–580.http://doi.org/10.1007/s10750-007-9032-2
Venegas-González, A., Juñent, F. ., Gutiérrez, A. G., & Tomazello Filho, M. (2018). Recent radial growth decline in response to increased drought conditions in the northernmost Nothofagus populations from South America. *Forest Ecology and Management*, *409*, 94–104. http://doi.org/10.1016/j.foreco.2017.11.006
Vergara, P. M., Pérez-Hernández, C. G., Hahn, I. J., & Soto, G. E. (2013). Deforestation in central Chile causes a rapid decline in landscape connectivity for a forest specialist bird species. *Ecological Research*, *28*(3), 481–492. http://doi.org/10.1007/s11284-013-1037-x
Vergara, P. M., & Simonetti, J. A. (2004). Avian responses to fragmentation of the Maulino forest in central Chile. *Oryx*, *38*(4), 383–388. http://doi.org/10.1017/S0030605304000742
Versveld, D. B., Le Maitre, D. C., & Chapman, R. A. (1998). *Alien Invading Plants and Water Resources in South Africa: a Preliminary Assessment*. Pretoria, South Africa: Water Research Commission.
Vihma, T., Screen, J., Tjernström, M., Newton, B., Zhang, X., Popova, V., Deser, C., Holland, M., & Prowse, T. (2016). The atmospheric role in the Arctic water cycle: A review on processes, past and future changes, and their impacts. *Journal of Geophysical Research: Biogeosciences*, *121*(3), 586–620. https://doi.org/10.1002/2015JG003132
Vila, I., Pardo, R., & Scott, S. (2007). Freshwater fishes of the Altiplano. *Aquatic Ecosystem Health & Management*, *10*(2), 201–211. http://doi.org/10.1080/14634980701351395
Vilela, B., & Villalobos, F. (2015). LetsR: a new R package for data handling and analysis in macroecology. *Methods in Ecology and Evolution*, *6*(10), 1229–1234.
Villagrán, C., & Castro, V. (2003). *Ciencia Indígena de los Andes del Norte de Chile*. Santiago. Chile: Editorial Universitaria.
Villagrán, C., & Hinojosa, L. F. (1997). History of the forests of southern South America .2. phytogeographical analysis. *Revista Chilena de Historia Natural*, *70*(2), 241–267.
Villagrán, C., Romo, M., & Castro, V. (2003). Etnobotánica del sur de los Andes de la primera Región de Chile: un enlace entre las culturas altiplánicas y las de quebradas altas del Loa superior. *Chungará*, *35*(1), 73–124.
Villalba, R., Boninsegna, J. A., Veblen, T. T., Schmelter, A., & Rubulis, S. (1997). Recent trends in tree-ring records from high elevation sites in the Andes of northern Patagonia. *Climatic Change*, *36*(3–4), 425–454. http://doi.org/10.1023/a:1005366317996
Villamagna, A. M., & Murphy, B. R. (2010). Ecological and socio-economic impacts of invasive water hyacinth (Eichhornia crassipes): a review. *Freshwater Biology*, *55*(2), 282–298. http://doi.org/10.1111/j.1365-2427.2009.02294.x
Villarino, S. H., Studdert, G. A., Baldassini, P., Cendoya, M. G., Ciuffoli, L., Mastrángelo, M., & Piñeiro, G. (2016). Deforestation impacts on soil organic carbon stocks in the Semiarid Chaco Region, Argentina. *Science of The Total Environment*, *575*, 1056–1065.

http://doi.org/10.1016/j.scitotenv.2016.09.175
Villegas, M., & Garitano-Zavala, Á. (2010). Bird community responses to different urban conditions in La Paz, Bolivia. *Urban Ecosystems*, *13*(3), 375–391. http://doi.org/10.1007/s11252-010-0126-7
Vitousek, P. M., Naylor, R., Crews, T., David, M. B., Drinkwater, L. E., Holland, E., Johnes, P. J., Katzenberger, J., Martinelli, L. A., Matson, P. A., Nziguheba, G., Ojima, D., Palm, C. A., Robertson, G. P., Sanchez, P. A., Townsend, A. R., & Zhang, F. S. (2009). Agriculture. Nutrient imbalances in agricultural development. *Science*, *324*, 1519–1520. https://doi.org/10.1126/science.1170261
Vitt, D. H. (2016). Peatlands of continental North America. In C. M. Finalyson, R. Milton, C. Prentice, & N. C. Davidson (Eds.), *The Wetland Book, II. Distribution, Description, and Conservation.* (pp. 1–6). Dordrecht, Netherlands: Springer.
Vo, Q. T., Kuenzer, C., Vo, Q. M., Moder, F., & Oppelt, N. (2012). Review of valuation methods for mangrove ecosystem services. *Ecological Indicators*, *23*, 431–446. http://doi.org/10.1016/j.ecolind.2012.04.022
Vörösmarty, C. J., Green, P., Salisbury, J., & Lammers, R. B. (2000). Global water resources: vulnerability from climate change and population growth. *Science*, *289*(5477), 284–288. http://doi.org/10.1126/science.289.5477.284
Vörösmarty, C. J., McIntyre, P. B., Gessner, M. O., Dudgeon, D., Prusevich, A., Green, P., Glidden, S., Bunn, S. E., Sullivan, C. A., Liermann, C. R., & Davies, P. M. (2010). Global threats to human water security and river biodiversity. *Nature*, *467*(7315), 555–561. https://doi.org/10.1038/nature09549
Voss, R. S., & Emmons, L. H. (1996). Mammalian diversity in neotropical lowland rainforests: a preliminary assessment. *Bulletin of the American Museum of Natural History*, *230*, 3–115. Retrieved from http://hdl.handle.net/2246/1671
Vuilleumier, F. (1985). Forest birds of Patagonia: ecological geography, speciation, endemism, and faunal history. *Ornithological Monographs*, *36*, 255–304. http://doi.org/10.2307/40168287
Vynne, C., Booth, R. K., & Wasser, S. K. (2014). Physiological implications of landscape use by free-ranging maned wolves (Chrysocyon brachyurus) in Brazil. *Journal of Mammalogy*, *95*(4), 696–706. http://doi.org/10.1644/12-MAMM-A-247
Walker, D. A., & Everett, K. R. (1987). Road dust and its environmental impact on Alaskan taiga and tundra. *Arctic and Alpine Research*, *19*(4), 479–489. http://doi.org/10.2307/1551414
Walker, J. S., Grimm, N. B., Briggs, J. M., Gries, C., & Dugan, L. (2009). Effects of urbanization on plant species diversity in central Arizona. *Frontiers in Ecology and the Environment*, *7*(9), 465–470. http://doi.org/10.1890/080084
Walker, M. D. (1995). Patterns and causes of Arctic plant community diversity. In F. S. Chapin III & C. Körner (Eds.), *Arctic and Alpine Biodiversity: Patterns, Causes and Ecosystem Consequences* (pp. 3–20). Berlin, Germany: Springer.
Walters, B. B., & Hansen, L. (2013). Farmed landscapes, trees and forest conservation in Saint Lucia (West Indies). *Environmental Conservation*, *40*(3), 211–221. http://doi.org/10.1017/S0376892912000446
Watling, L., & Norse, E. A. (1998). Disturbance of the seabed by mobile fishing gear: a comparison to forest clearcutting. *Conservation Biology*, *12*(6), 1180–1197. http://doi.org/10.1046/j.1523-1739.1998.0120061180.x
Watson, J. E. M., Dudley, N., Segan, D. B., & Hockings, M. (2014). The performance and potential of protected areas. *Nature*, *515*(7525), 67–73. http://doi.org/10.1038/nature13947
Webb, C. O., Ackerly, D. D., McPeek, M. A., & Donoghue, M. J. (2002). Phylogenies and community ecology. *Annual Review of Ecology and Systematics*, *33*(1), 475–505. http://doi.org/10.1146/annurev.ecolsys.33.010802.150448
Weed, A. S., Ayres, M. P., & Hicke, J. A. (2013). Consequences of climate change for biotic disturbances in North American forests. *Ecological Monographs*, *83*(4), 441–470. http://doi.org/10.1890/13-0160.1
Wege, D. C., Ryan, D., Varty, N., Anadon-Irizarry, V., & Perez-Leroux, A. (2010). Ecosystem Profile: The Caribbean Islands Biodiversity Hotspot. Arlington, USA: Critical Ecosystem Partnership Fund.
Wehrtmann, I. S., Ramírez, A., & Pérez-Reyes, O. (2016). Freshwater decapod diversity and conservation in Central America and the Caribbean. In T. Kawai & N. Cumberlidge (Eds.), *A Global Overview of the*

Conservation of Freshwater Decapod Crustaceans (pp. 267–301). Cham, Switzerlan: Springer.

Weigelt, A., Weisser, W. W., Buchmann, N., & Scherer-Lorenzen, M. (2009). Biodiversity for multifunctional grasslands: equal productivity in high-diversity low-input and low-diversity high-input systems. *Biogeosciences*, *6*(8), 1695–1706. http://doi.org/10.5194/bgd-6-3187-2009

Wester, L. (1991). Invasions and extinctions on Masatierra (Juan Fernandez Islands): a review of early historical evidence. *Journal of Historical Geography*, *17*(1), 18–34. http://doi.org/10.1016/0305-7488(91)90003-E

Westerling, A. L., Hidalgo, H. G., Cayan, D. R., & Swetnam, T. W. (2006). Warming and earlier spring increase western U.S. forest wildfire activity. *Science*, *313*(5789), 940–943.http://doi.org/10.1126/science.1128834

Westoby, M. (1998). A leaf-height-seed (LHS) plant ecology strategy scheme. *Plant and Soil*, *199*(2), 213–227. http://doi.org/10.1023/A:1004327224729

Wetmore, A., Friedmann, H., Amadon, D., Lincoln, F. C., Lowery, G. H., Miller, A. H., Peters, J. L., Pitelka, F. A., van Rossem, A. J., van Tyle, J., & Zimmer, J. T. (1957). *Check-list of North American Birds, 5th Edition*. Baltimore, USA: American Ornithologists' Union.

Wheeler, J. C. (2017). Evolución y domesticación en los camélidos sudamericanos. In A. Casas, J. Torres-Guevara, & F. Parra (Eds.), *Domesticación en el Continente Americano Vol. 2. Investigación para el Manejo Sustentable de Recursos Genéticos en el Nuevo Mundo* (pp. 193–216). México D.F., México and Lima, Perú: Universidad Nacional Autónoma de México/Universidad Nacional Agraria La Molina/CONACYT.

Whitfield, S. M., Lips, K. R., & Donnelly, M. A. (2016). Amphibian decline and conservation in Central America. *Copeia*, *104*(2), 351–379. http://doi.org/10.1643/CH-15-300

Wiegmann, S. M., & Waller, D. M. (2006). Fifty years of change in northern upland forest understories: identity and traits of "winner" and "loser" plant species. *Biological Conservation*, *129*(1), 109–123. http://doi.org/10.1016/j.biocon.2005.10.027

Wierzchos, J., Ascaso, C., & McKay, C. P. (2006). Endolithic cyanobacteria in halite rocks from the hyperarid core of the Atacama Desert. *Astrobiology*, *6*(3), 415–422. http://doi.org/10.1089/ast.2006.6.415

Wiig, Ø., Amstrup, S., Atwood, T., Laidre, K., Lunn, N., Obbard, M., Regehr, E., & Thiemann, G. (2015). *Ursus maritimus*. *The IUCN Red List of Threatened Species 2015:* Retrieved from http://dx.doi.org/10.2305/IUCN.UK.2015-4.RLTS.T22823A14871490.en

Will-Wolf, S., & Stearns, F. (1999). Dry soil oak savanna in the Great Lakes region. In R. C. Anderson, J. S. Fralish, & J. M. Baskin (Eds.), *Savannas, Barrens, and Rock Outcrop Plant Communities of North America* (pp. 135–154). Cambridge, UK: Cambridge University Press.

Williams, A. P., Allen, C. D., Macalady, A. K., Griffin, D., Woodhouse, C. A., Meko, D. M., Swetnam, T. W., Rauscher, S. A., Seager, R., Grissino-Mayer, H. D., Dean, J. S., Cook, E. R., Gangodagamage, C., Cai, M., & McDowell, N. G. (2013). Temperature as a potent driver of regional forest drought stress and tree mortality. *Nature Climate Change*, *3*(3), 292–297. https://doi.org/10.1038/nclimate1693

Williams, A. P., Seager, R., Abatzoglou, J. T., Cook, B. I., Smerdon, J. E., & Cook, E. R. (2015). Contribution of anthropogenic warming to California drought during 2012–2014. *Geophysical Research Letters*, *42*(16), 6819–6828. http://doi.org/10.1002/2015GL064924

Williams, J. D., Warren Jr, M. L., Cummings, K. S., Harris, J. L., & Neves, R. J. (1993). Conservation status of freshwater mussels of the United States and Canada. *Fisheries*, *18*(9), 6–22. http://doi.org/10.1577/1548-8446(1993)018<0006:CSOFMO>2.0.CO;2

Willis, K. J. (Ed.). (2016). *State of the World´s Plants*. London, UK: Royal Botanic Gardens, Kew. Retrieved from https://stateoftheworldsplants.com/2016/

Wilsey, B. J., Martin, L. M., & Polley, H. W. (2005). Predicting plant extinction based on species-area curves in prairie fragments with high beta richness. *Conservation Biology*, *19*(6), 1835–1841.http://doi.org/10.1111/j.1523-1739.2005.00250.x

Wilsey, B. J., & Polley, H. W. (2004). Realistically low species evenness does not alter grassland species-richness-productivity relationships. *Ecology*, *85*(10), 2693–2700. http://doi.org/10.1890/04-0245

Wilson, B. S., Koenig, S. E., van Veen, R., Miersma, E., & Rudolph, D. C. (2011). Cane toads a threat to West Indian wildlife: mortality of Jamaican boas attributable to toad ingestion. *Biological Invasions*, *13*(1), 55–60. http://doi.org/10.1007/s10530-010-9787-7

Wilson, T. S., Sleeter, B. M., & Cameron, D. R. (2016). Future land-use related water demand in California. *Environmental Research Letters*, *11*(5), 1–12. http://doi.org/10.1088/1748-9326/11/5/054018

Winograd, M., Farrow, A., Aguilar, M., Clavijo, L. A., Debouck, D. G., Jones, P. G., Hyman, G., Lema, G., Nelson, A. N., Holmann, F. J., Tohme, M. J., Toro, C. O., & Voyset, V. O. (1999). *Agroecosystem Assessment for Latin America: Agriculture Extent, Production Systems and Agrobiodiversity*. Cali, Colombia: Centro Internacional de Agricultura Tropical (CIAT).

Wolf, A., Zimmerman, N. B., Anderegg, W. R. L., Busby, P. E., & Christensen, J. (2016). Altitudinal shifts of the native and introduced flora of California in the context of 20th-century warming. *Global Ecology and Biogeography*, *25*(4), 418–429. http://doi.org/10.1111/geb.12423

World Cities Culture Forum. (n.d.). % of public green space (parks and gardens). Retrieved from http://www.worldcitiescultureforum.com/data/of-public-green-space-parks-and-gardens

World Database of Key Biodiversity Areas. (n.d.). Retrieved from http://www.keybiodiversityareas.org

World Wildlife Fund (WWF). (2016). *Plowprint Report*. Retrieved from https://www.worldwildlife.org/projects/plowprint-report

World Wildlife Fund (WWF). (2017a). *2017 Plowprint Report*. Bozeman, USA. Retrieved from https://www.worldwildlife.org/projects/plowprint-report

World Wildlife Fund (WWF). (2017b). Northern South America: Northeastern Brazil. Retrieved from https://www.worldwildlife.org/ecoregions/nt1304

Worm, B., Hilborn, R., Baum, J. K., Branch, T. A., Collie, J. S., Costello, C., … Zeller, D. (2009). Rebuilding global fisheries. *Science*, *325*(5940), 578–585. http://doi.org/10.1126/science.1173146

Wright, I. J., Reich, P. B., Westoby, M., Ackerly, D. D., Baruch, Z., Bongers, F., Cavender-Bares, J., Chapin, T., Cornelissen, J. H. C., Diemer, M., Flexas, J., Garnier, E., Groom, P. K., Gulias, J., Hikosaka, K., Lamont, B. B., Lee, T., Lee, W., Lusk, C., Midgley, J. J., Navas, M.-L., Niinemets, Ü., Oleksyn, J., Osada, N., Poorter, H., Poot, P., Prior, L., Pyankov, V. I., Roumet, C., Thomas, S. C., Tjoelker, M. G., Veneklaas, E. J., & Villar, R. (2004). The worldwide leaf economics spectrum. *Nature*, *428*(6985), 821–827. https://doi.org/10.1038/nature02403

Wright, J. A., DiNicola, A., & Gaitan, E. (2000). Latin American forest plantations: opportunities for carbon sequestration, economic development, and financial returns. *Journal of Forestry*, *98*(9), 20–23. Retrieved from http://www.scopus.com/inward/record.url?eid=2-s2.0-0034283558&partnerID=40&md5=0cc1b0bb502bf3c27297f4b0452c9f0a

Wrigley de Basanta, D., Lado, C., Estrada-Torres, A., & Stephenson, S. L. (2010). Biodiversity of myxomycetes in subantarctic forests of Patagonia and Tierra del Fuego, Argentina. *Nova Hedwigia*, *90*(1–2), 45–79. http://doi.org/10.1127/0029-5035/2010/0090-0045

Young, B. E., Lips, K. R., Reaser, J. K., Ibáñez, R., Salas, A. W., Cedeño, J. R., Coloma, L. A., Ron, S., La Marca, E., Meyer, J. R., Muñoz, A., Bolaños, F., Chaves, G., & Romo, D. (2001). Population declines and priorities for amphibian conservation in Latin America. *Conservation Biology*, *15*(5), 1213–1223. https://doi.org/10.1111/j.1523-1739.2001.00218.x

Young, C. N., Carlson, J., Hutchinson, M., Kobayashi, D., McCandless, C., Miller, M. H., Teo, S., & T. Warren. (2016). *Status review report: common thresher shark (Alopias vulpinus) and bigeye thresher shark (Alopias superciliosus). Final Report to National Marine Fisheries Service, Office of Protected Resources*.

Young, K. L., Lafrenière, M. J., Lamoureux, S. F., Abnizova, A., & Miller, E. A. (2015). Recent multi-year streamflow regimes and water budgets of hillslope catchments in the Canadian High Arctic: evaluation and comparison to other small Arctic watershed studies. *Hydrology Research*, *46*(4), 533–550. http://doi.org/10.2166/nh.2014.004

Young, O. R., & Chapin III, F. S. (1995). Anthropogenic impacts on biodiversity in the Arctic. In F. S. Chapin III & C. Körner (Eds.), *Arctic and Alpine Biodiversity: Patterns, Causes and Ecosystem Consequences* (pp. 183–196). Berlin, Germany: Springer.

Zak, M. R., Cabido, M., & Hodgson, J. G. (2004). Do subtropical seasonal forests in the Gran Chaco, Argentina, have a future? *Biological Conservation*, *120*(4), 589–598. http://doi.org/10.1016/j.biocon.2004.03.034

Zamora Nasca, L., Montti, L., Grau, R., & Paolini, L. (2014). Effects of glossy privet's invasion on the water dynamics of the Argentinean Yungas forest. *Bosque*, *35*(2), 195–205.

http://doi.org/10.4067/S0717-92002014000200007
Zamorano-Elgueta, C., Rey Benayas, J. M., Cayuela, L., Hantson, S., & Armenteras, D. (2015). Native forest replacement by exotic plantations in southern Chile (1985-2011) and partial compensation by natural regeneration. *Forest Ecology and Management*, *345*, 10–20. http://doi.org/10.1016/j.foreco.2015.02.025

Zanne, A. E., Tank, D. C., Cornwell, W. K., Eastman, J. M., Smith, S. A., FitzJohn, R. G., McGlinn, D. J., O'Meara, B. C., Moles, A. T., Reich, P. B., Royer, D. L., Soltis, D. E., Stevens, P. F., Westoby, M., Wright, I. J., Aarssen, L., Bertin, R. I., Calaminus, A., Govaerts, R., Hemmings, F., Leishman, M. R., Oleksyn, J., Soltis, P. S., Swenson, N. G., Warman, L., & Beaulieu, J. M. (2013). Three keys to the radiation of angiosperms into freezing environments. *Nature*, *506*, 89–92. https://doi.org/10.1038/nature12872

Zarazúa, M. (2016). Del guajolote a las chicatanas. Uso, manejo y domesticación de recursos genéticos animales en Mesoamérica. In A. Casas, J. Torres-Guevara, & F. Parra (Eds.), *Domesticación en el Continente Americano Vol. 1. Investigación Manejo de Biodiversidad y Evolución Dirigida por las Culturas del Nuevo Mundo* (pp. 283–316). México D.F., México and: Universidad Nacional Autónoma de México/Universidad Nacional Agraria La Molina.

Zarin, D. J., Harris, N. L., Baccini, A., Aksenov, D., Hansen, M. C., Azevedo-Ramos, C., Azevedo, T., Margono, B. A., Alencar, A. C., Gabris, C., Allegretti, A., Potapov, P., Farina, M., Walker, W. S., Shevade, V. S., Loboda, T. V, Turubanova, S., & Tyukavina, A. (2016). Can carbon emissions from tropical deforestation drop by 50% in 5 years? *Global Change Biology*, *22*(4), 1336–1347. https://doi.org/10.1111/gcb.13153

Zavaleta, E., Tershy, B., Harrison, S., Borker, A., Sinervo, B., Cornelisse, T., Li, C., Spatz, D., & Croll, D. (2016). Biodiversity. In H. Mooney and E. Zavaleta (Ed.), *Ecosystems of California* (pp. 187–212). Oakland, USA: University of California Press.

Zedler, J. B., & Kercher, S. (2005). Wetland resources: Status, trends, ecosystem services, and restorability. *Annual Review of Environment and Resources*, *30*, 39–74. http://doi.org/10.1146/annurev.energy.30.050504.144248

Zedler, P. H. (1995). Fire frequency in southern California shrublands: biological effects and management options. In J. E. Keeley & T. Scott (Eds.), *Brushfires in California: Ecology and Resource Management* (pp. 101–112). Fairfield, USA: International Association of Wildland Fire.

Zolla, C., & Zolla, E. (2004). *Los Pueblos Indígenas de México, 100 Preguntas*. México D.F., México: Universidad Nacional Autónoma de México.

4 Chapter 4: Direct and indirect drivers of change in biodiversity and nature’s contributions to people

Coordinating Lead Authors:
Mercedes Bustamante (Brazil), Eileen H. Helmer (USA), Steven Schill (USA)

Lead Authors:
Jayne Belnap (USA), Laura K. Brown (Canada), Ernesto Brugnoli (Uruguay), Jana E. Compton (USA), Richard H. Coupe (USA), Marcello Hernández-Blanco (Costa Rica), Forest Isbell (USA), Julie Lockwood (USA), Juan Pablo Lozoya Azcárate (Uruguay), David McGuire (USA), Anibal Pauchard (Chile), Ramon Pichs-Madruga (Cuba), Ricardo Ribeiro Rodrigues (Brazil), Gerardo Arturo Sanchez-Azofeifa (Costa Rica/Canada), Alvaro Soutullo (Uruguay), Avelino Suarez (Cuba), Elizabeth Troutt (Canada).

Fellow:
Laura Thompson (USA).

Contributing Authors:
Robin Abell (USA), Lorenzo Alvarez-Filip (Mexico), Christopher B. Anderson (Argentina/USA), Adriana De Palma (UK), Arturo Dominici (Panama), Javier Godar (Spain), Gladys Hernandez (Cuba), Myanna Lahsen (Brazil), Marília Cunha-Lignon (Brazil), Frank Muller-Karger (USA), Laura Nahuelhual (Chile), Judith A. Perlinger (USA), Helder Lima Queiroz (Brazil), Carla R. G. Reis (Brazil), Carmen Revenga (USA), Jeremy Rude (USA), Dalia Salabarria (Cuba), Jennifer J. Swenson (USA), Noel R. Urban (USA).

Review Editors:
Pedro Laterra (Argentina), Carlos Eduardo Young (Brazil)

To be cited as:
Bustamante, M., Helmer, E. H., Schill, S., Belnap, J., Brown, L. K., Brugnoli, E., Compton, J. E., Coupe, R. H., Hernández-Blanco, M., Isbell, F., Lockwood, J., Lozoya Ascárate., J. P., McGuire, D., Pauchard, A., Pichs-Madruga, R., Rodrigues, R. R., Sanchez- Azofeifa, G. A., Soutullo, A., Suarez, A., Troutt, E., and Thompson, L. Chapter 4: Direct and indirect drivers of change in biodiversity and nature’s contributions to people. In IPBES (2018): The IPBES regional assessment report on biodiversity and ecosystem services for the Americas. Rice, J., Seixas, C. S., Zaccagnini, M. E., Bedoya-Gaitán, M., and Valderrama, N. (eds.). Secretariat of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services, Bonn, Germany, pp. xx-xx

Table of contents Chapter 4

4.1 Executive summary

1. **The most important indirect anthropogenic drivers of changes in nature, nature's contributions to people and good quality of life include unsustainable patterns of economic growth (including issues related to international trade and finances); population and demographic trends; weaknesses in the governance systems and inequity (*well established*).** Increasing human demand for food, water, and energy caused by increases in population, per capita Gross Domestic Product and international trade have had negative consequences for nature and many regulating and non-material nature's contributions to people.

2. **Social inequity is a concern with adverse implications for nature, nature's contributions to people and good quality of life (*well established*). When the United Nations Development Program Human Development Index is adjusted for inequality, it is 22 per cent lower in Latin American and Caribbean countries and 11.1 per cent lower in North America {4.3.6}.** Seventy-two million people escaped income-poverty from 2003-2013 in Latin America; however, around 26.9 per cent of the Latin American population still lived in poverty in 2012: 40.6 per cent in Mesoamerica and 21 per cent in South America {4.3.6}. In many cases, poor people in the Americas tend to increase the pressures on nature merely to survive, while on the other hand, there is high per capita consumption of natural resources in affluent segments of the population.

3. **Economic growth (measured as Gross Domestic Product growth and Gross Domestic Product per capita) and international trade are major drivers of natural resource consumption in the Americas. Economic growth and trade can positively or negatively impact biodiversity and nature's contributions to people, but currently, on balance, they adversely impact biodiversity and nature's contributions to people when environmental and social development goals are insufficiently accounted for (*well established*).** Positive impacts of economic growth and international trade may include a stronger economy and increased employment, and social and environmental investments such as biodiversity protection. Negative impacts of economic growth include unsustainable conversion, use and exploitation of terrestrial, freshwater and marine ecosystems and resources, which threaten biodiversity and degrade nature's contributions to people by reducing species abundances below self-sustaining levels and by disrupting key ecosystem functions {4.6}. The Americas generates around 18 per cent of world exports, with 70 per cent of this from North America. The Latin American and Caribbean contributions to world exports is 5.4 per cent, and natural resource governance is strongly influenced by having economies dominated by commodity exports. Natural resources (oil, minerals, and agriculture) contribute more than 50 per cent to these Latin America and the Caribbean exports {4.3.3}. Globalization has catalyzed rapid growth of international trade and become an important motor for regional development, but it has also disconnected places of production, transformation and consumption of land-based products. This decoupling places significant challenges for socio-environmental governance and regulatory implementation for sectors rapidly changing in response to increases in the global demand for food, feed and fiber. Consequently, natural resource use policies often come into place only after fundamental shifts in the land-use system are already underway, and interventions have become costly and have limited influence {4.6}.

4. **Weaknesses in the governance systems and institutional frameworks in the Americas have had adverse implications for nature, nature's contributions to people and good quality of life in the Americas (*well established*).** In most countries in the region centralized modes of governance still prevail where decision-making regarding Nature and nature's contributions to people in reality falls on the State. Centralized command and control measures nonetheless, such as the establishment of protected areas, continue to be a pillar of biodiversity conservation. Significant progress has been made to include other actors and new hybrid governance modes such as public-private certification schemes or payment for ecosystem services, which are in line with the rising role of markets in environmental governance. These transformations from centralized to descentralized forms, however, have led to significant socioenvironmental conflicts in the region {4.3.1}.

5. **Value systems in the Americas differ among cultural groups and identities across the whole region**

and shape governance systems, in particular the ways of addressing development policies, land tenure and indigenous rights, and strongly influence decisions on land use and natural resources exploitation in the different subregions (*well established*). Indigenous and traditional peoples throughout the Americas have developed many different socio-economic systems (nationally and locally). Indigenous and local knowledge are expressions of social articulations that can positively influence biodiversity and ecosystem services. While cases that conservation of biodiversity and nature's benefits to people are related to empowerment of indigenous and traditional communities are emerging in the region (for example, the role of indigenous land on deforestation control in tropical forests of South America), weak and less participatory governance systems are associated with cases of conflicts in managing land and natural resources in all of the Americas subregions (for example, conflicts related to infrastructure building in indigenous lands) {4.3.1, 4.3.6}.

6. **Habitat conversion, fragmentation and overexploitation/overharvesting are resulting in a loss of biodiversity and a loss of nature's contributions to people in all ecosystems. Habitat degradation due to land conversion and agricultural intensification; wetland drainage and conversion; urbanization and other new infrastructure, and resource extraction is the largest threat to fresh water, marine and terrestrial biodiversity and nature's contributions to people in the Americas (*well established*).** The resulting changes in terrestrial, freshwater and marine environments are interrelated and often lead to changes in biogeochemical cycles, pollution of ecosystems and eutrophication, and biological invasions, which are at the same time significant direct drivers of change in the region (*well established*). The expansion and intensification of agriculture and livestock production in the Americas are decreasing the area of and altering natural ecosystems (*well established*) {4.4.1}. Related changes include shifting drainage patterns (affecting infiltration and runoff), water quality degradation, soil disturbance, habitat loss, and release of chemicals that can be toxic to biota and human populations. Nitrogen and phosphorus fertilizer use have greatly contributed to increases in the amount of available nitrogen and phosphorus in the environment, doubling available nitrogen, for example, with negative consequences for ecosystem function, and air, soil and water quality {4.4.2}, including major contributions to coastal and freshwater oxygen depletion. Land-use changes, road and trail construction, waterways and domestic animals are common dispersal routes for invasive species (*well established*) {4.4.4}. Habitat conversion also decreases connectivity among, and diversity within, remaining fragments of natural ecosystems (*well established*). Wildlife, fisheries, and people, including many indigenous peoples, are exposed to residual pollution in the environment. Mining for trace metal ores and coal has left lasting legacies of toxic pollution across the region {4.4.2} (*well established*). Although unsustainable management of natural resources are threatening biodiversity and degrading nature's contributions to people by reducing populations below natural self-sustaining levels and disrupting ecosystem functions {4.4.5}, some sustainable practices have been identified and used in terrestrial and aquatic environments.

7. **Rapid urbanization is a key driver of loss of biodiversity and nature's contributions to people, but the nature and the magnitude of impacts vary substantially among subregions of the Americas (*established but incomplete*).** The Americas region is highly urbanized, with about 80 per cent of the region's population residing in urban settings {4.3.5}. Although urban population impacts depend on consumption patterns and lifestyles, which vary considerably from one subregion to another, in all subregions a large number of ecosystems have been affected. Urbanization driven by growing populations and internal migration acts as an indirect driver of land-use change through linear infrastructures. In Latin America and the Caribbean, 12 per cent of the urban population and 36 per cent of rural population do not have access to improved sanitation facilities, and only 50 per cent of the population in Latin America is connected to sewerage. The poor systematic waste management in Latin America and the Caribbean implies pollution of inland waters and coastal areas {4.4.2} affecting biodiversity and human health.

8. **Carbon dioxide emissions from fossil fuel production continue to increase, increasing 29 per cent from 2000 to 2008. The combustion of fossil fuels is not only the primary source of anthropogenic greenhouse gases that cause human-induced climate change, but fossil fuel combustion itself is also a major source of pollution adversely impacting most terrestrial and marine ecosystems and human**

health {4.4.2} (*well established*). Air pollution (especially particulates, ozone, mercury, and carcinogens) causes significant adverse health effects on infants, adults and biodiversity (*well established*), and carbon dioxide emissions cause ocean acidification. For example, the combustion of fossil fuels account for 25 per cent of the direct anthropogenic mercury emissions that are increasing the mercury burden of polar and subpolar wildlife and indigenous people with diets dominated by fish, eggs of fish-eating birds, and marine mammals, affecting wildlife reproduction and infant nervous systems. Ocean acidification from increased atmospheric carbon dioxide is increasing and is already impacting major components of the Pacific Ocean food web and contributing to a Caribbean-wide flattening of coral reefs. If current trends continue, coral reef systems will be further adversely affected. Ocean temperatures have become warmer, and together with nutrient run-off, are contributing to increasing ocean deoxygenation. Fossil fuel combustion also contributes to human-caused atmospheric nitrogen deposition, being responsible for 16 per cent of anthropogenic creations of reactive nitrogen, which shifts the species composition of ecosystems and makes groundwater toxic. Fossil fuel related nitrogen emissions have declined in North America.

9. **Marine plastic pollution is increasing, and it is expected to exacerbate stresses on the marine food web from warming temperatures, acidification and overexploitation (*establisehd but incomplete*).** In 2010, globally and from land-based sources alone, five to 13 million metric tons of plastic pollution entered the ocean. Two countries of the Americas are among the 20 top polluters. The environmental implications of microplastics at sea are still largely unknown, however the number of marine species known to be affected by this contaminant has gone from 247 to 680 {4.4.2}. New evidence indicates microplastics have a complex effect on marine life and are is transferred up the food chain to people. Impacts on marine wildlife include entanglement, ingestion, death and contamination to a wide variety of species.

10. **Human induced climate change caused by the emissions of greenhouse gases is becoming an increasingly more important direct driver, amplifying the impacts of other drivers (i.e. habitat degradation, pollution, invasive species and overexploitation) through changes in temperature, precipitation and frequency of extreme events and other variables (*well- established*).** Climate change has, and will continue to, adversely affect biodiversity at the genetic, species and ecosystem level. The majority of ecosystems in the Americas have already experienced increased mean and extreme temperatures and/or precipitation which have, for example, caused changes in species distributions and ecosystem boundaries, and caused mountain glaciers to retreat. However, the interaction between these direct impacts and other direct and indirect drivers are increasing vulnerability of sensitive ecosystems through the interaction of warming temperatures and pollution, as in the example of coral reefs in the Caribbean. The main impacts on terrestrial, freshwater and marine species are the shift in their geographic ranges, and changes in seasonal activities, migration patterns and abundances. Species affected by other drivers are less resilient to climate change and therefore have a high extinction risk.

11. **Although most ecosystems in the America's continue to be degraded, increases in conservation (e.g. protected areas), and in ecological restoration, are having positive effects. Ecological restoration significantly speeds up ecosystem recovery in some cases (*well established*), but costs can be significant, and full reversal of the adverse impacts of humans on nature is unlikely to be achievable (*well established*).** Evidence from different subregions indicates that structure and functionality of ecosystems recover faster than species richness (particularly in species-rich biomes). Non-material contributions of naature to people may not be restored for some people {4.4.1}.

12. **In spite of the pressures of drivers of change on nature and nature's contributions to people, there are management and policy options that can affect the drivers of change in order to mitigate, and most importantly, to avoid, impacts on different ecosystems (*establisehd but incomplete*)**. However, given the current status and trends of drivers, meeting the Aichi targets and Sustainable Development Goals will require stronger and more effective efforts on the parts of the countries across the region. These options and their implementation are context dependent and strongly influenced by values, governance and institutions {4.7}. Such conditions vary substantially across the Americas in relation to

social and economic inequity.

4.2 Introduction

The Americas encompass seven magadiverse countries (one in North America, one in Mesoamerica and five in South America) of the 17 in the world (see Chapter 1 for more details). However, the degradation of critical ecosystems and loss of biodiversity in the region threaten human well-being by impacting important ecosystem functions and services, like clean air and water, flood and climate control, and soil regeneration, as well as food, medicines and raw materials (see Chapter 2 for more details).

As a function of the pressure on natural ecosystems, the Americas contain 10 of the 36 world biodiversity hotspots, i.e. areas with high biodiversity facing extreme threats and that have lost at least 70 percent of their original habitat: 1. California floristic province (USA), 2. North American coastal plain (USA), 3. madrean pine-oak woodlands (USA and Mexico), 4. Mesoamerica, 5. Caribbean islands, 6. Atlantic forest (Brazil), 7. Cerrado (Brazil), 8. Chilean winter rainfall-Valdivian forests (Chile), 9. Tumbes-Chocó-Magdalena (Colombia) and 10. Tropical Andes (Marchese, 2015, http://www.cepf.net/resources/hotspots/).

Environmental problems are also wide-ranging and vary between and within nations. Negative environmental trends are observed throughout the region, which are to a large extent the result of long historical patterns of growth induced by non-sustainable consumption. A significant feature of these environmental problems is that they are often shared among countries, including climate change and disaster risk management, sustainable management of land and ecosystems, water resources management, sustainable energy management, good governance for inclusive and sustainable development, such that regional cooperation is needed to tackle them (UNEP, 2016).

Social and economic inequality and weak environmental governance are common features in the Americas that are intricately linked with a deteriorating environment. Environmental and climate change issues are gaining weight regionally, but unsustainable development models still predominate, with significant consequences for the environment and human well-being. Lack of security and equity in accessing basic resources (like land ownership or user rights, access to the natural commons and fundamental ecosystem services) do not provide incentives for sustainable management or increased efficiency. However, sustainable use might provide an opportunity to improve welfare for the people (UNEP, 2016)

Given the importance of the Americas' biodiversity and ecosystem services for human well-being (see Chapters 2 and 3 for more details), this chapter explores key drivers of changes in biodiversity and ecosystem services in the region. These include indirect and direct anthropogenic drivers as well as direct natural drivers.

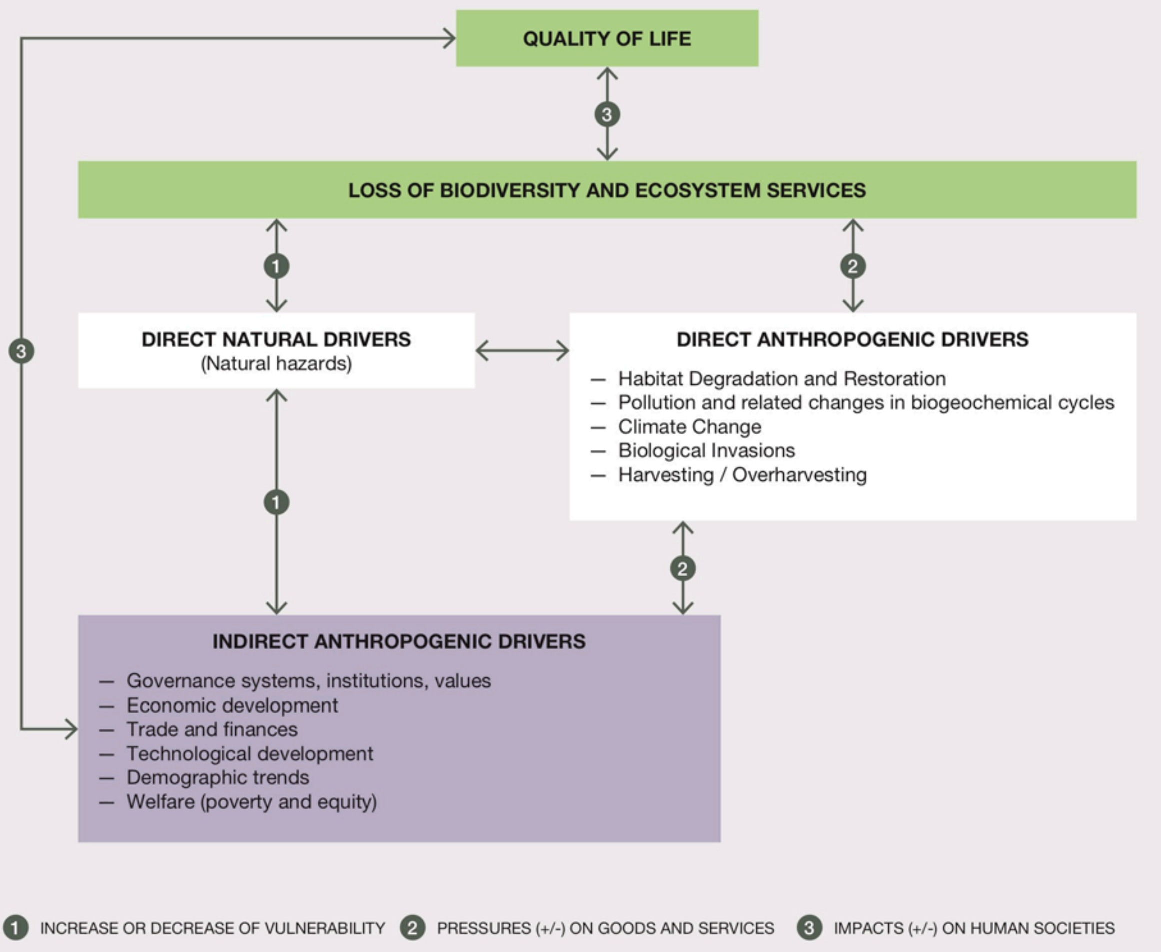
A range of drivers, including environmental change and human uses of resources, induce changes in biodiversity and ecosystems. A driver is any natural or human-induced factor that directly or indirectly causes a change. A direct driver unequivocally influences ecosystem processes. An indirect driver operates more diffusely, by altering one or more direct drivers. Box 4.1 summarizes the definitions on drivers included in the Intergovernmental Platform on Biodiversity and Ecosystem Services (IPBES) conceptual framework (Decision IPBES-2/4, available on http://www.ipbes.net).

The drivers examined in this chapter are primarily anthropogenic. Indirect anthropogenic drivers are aspects and patterns of human organization and socioeconomic activity (section 4.3) that produce aggregate outcomes that in turn bring about changes in biodiversity and ecosystem services. Direct anthropogenic drivers (section 4.4) are the aggregate outcomes, such as habitat change, pollution or climate change, from the indirect anthropogenic drivers that yield those changes. Direct natural drivers also produce changes in biodiversity and ecosystem services, and are thus also presented briefly in this chapter (section 4.5).

As **¡Error! No se encuentra el origen de la referencia.** shows, the indirect and direct anthropogenic drivers are significantly interrelated. Even though sections 4.3, 4.4, and 4.5 describe these drivers sequentially and distinctly, important interactions are also presented in the specific sections (indicated in bold along the text). These interactions will be synthesized in section 4.6, while the effects of indirect drivers on direct drivers are further discussed in section 4.7. Section 4.8 provides a starting indication of where gaps in current scientific knowledge lie. The gaps in knowledge point to areas where data remain insufficient and to areas where further data collection and scientific inquiry and analysis are needed to produce a stronger understanding of the links between indirect and direct anthropogenic drivers, changes in biodiversity and ecosystem services, and human well-being.

Lastly, section 4.9 contains supplementary material that enrich the chapter by displaying additional content that add detail, background, or context by resources such as case studies, figures and tables.

Figure 4 1 Interactions among multiple indirect and direct drivers of biodiversity loss and ecosystem changes. Source: own representation.



Box 4.1. Definitions of drivers of change of nature´s contributions to people and good quality of life, and partial representation of the IPBES conceptual framework according to IPBES Decision 2-4

Drivers of change refers to all those external factors that affect nature, anthropogenic assets, nature's contributions to people and a good quality of life. They include institutions and governance systems and other indirect drivers and direct drivers (both natural and anthropogenic).

Institutions and governance systems and other indirect drivers are the ways in which societies organize themselves, and the resulting influences on other components. They are the underlying causes of environmental change that are exogenous to the ecosystem in question. Because of their central role, influencing all aspects of human relationships with nature, these are key levers for decision-making. Institutions encompass all formal and informal interactions among stakeholders and social structures that determine how decisions are taken and implemented, how power is exercised, and how responsibilities are distributed. Institutions determine, to various degrees, the access to, and the control, allocation and distribution of components of nature and anthropogenic assets and their benefits to people.

Direct drivers, both natural and anthropogenic, affect nature directly.

Natural drivers are those that are not the result of human activities and are beyond human control. These include earthquakes, volcanic eruptions and tsunamis, extreme weather or ocean-related events such as prolonged drought or cold periods, tropical cyclones and floods, the El Niño/La Niña Southern Oscillation and extreme tidal events.

The direct anthropogenic drivers are those that are the result of human decisions, namely, of institutions and governance systems and other indirect drivers. Anthropogenic drivers include habitat conversion, e.g. degradation of land and aquatic habitats, deforestation and afforestation, exploitation of wild populations, climate change, pollution of soil, water and air and species introductions. Some of these drivers, such as pollution, can have negative impacts on nature; others, as in the case of habitat restoration, or the introduction of a natural enemy to combat invasive species, can have positive effects. Institutions and governance systems and other indirect drivers affect all elements and are the root causes of the direct anthropogenic drivers that directly affect nature and also affect the interactions and balance between nature and human assets in the co-production of nature's benefits to people

Anthropogenic assets refer to built-up infrastructure, health facilities, knowledge (including indigenous and local knowledge systems and technical or scientific knowledge, as well as formal and non-formal education), technology (both physical objects and procedures), as financial assets, among others. Direct drivers also affect anthropogenic assets and in addition, anthropogenic assets directly affect the possibility of leading a good life through the provision of and access to material wealth, shelter, health, education, satisfactory human relationships, freedom of choice and action, and sense of cultural identity and security. These linkages are acknowledged but not addressed in depth because they are not the main focus of IPBES.

4.3 Indirect anthropogenic drivers

Indirect drivers (also referred as underlying factors) play a major role in influencing direct drivers (proximate causes) of changes in nature, nature's contributions to people and good quality of life in different spatial and temporal scales, involving "anthropogenic assets" (encompassing infrastructure, knowledge systems, including indigenous and local knowledge (ILK), technology and financial assets, among others). Considering the concept and the nature of complex ecological systems, the role of indirect drivers is an integral aspect of natural resource use assessments, and needs to be considered to explain and study past and ongoing processes as well as for scenario development and subsequent analysis (IPBES, 2016).

The indirect anthropogenic drivers can be classified according to the origin of the driver, which for instance can be fed by predominantly local processes, like for example poor local governance and corruption. It is widely recognized that globalization in recent decades has led to "spatial decoupling of the local land uses from their most important driving forces" (Reenberg et al., 2010). This recent observation has led to the establishment of the teleconnection framework (Friis et al., 2015; Kastner et al., 2015). For instance, changes in land systems at various spatial scales are influenced by long-distance flows of capital, energy, traded products, people and information. While locally driven processes have been studied for decades using perspectives from different disciplines (demography, anthropology, political economy), teleconnections have been assessed only in the last decade. Furthermore, it is only recently that the teleconnection framework has given birth to the concept of telecoupling (Liu et al., 2013), which considers also the multiple feedbacks and teleconnected interactions in both socioeconomic and environmental terms. For example, climate risks may be transmitted to a region via trade networks, but also through migration flows into that region that can be triggered by climate risks elsewhere. In both cases local socio-economic conditions in that region are affected, and therefore its natural resource management. The complexity and multi-layered nature of these interactions hampers the design and implementation of governance measures. However, at the same time it may also allow the participation of a number of distal actors and processes, opening space for mobilizing resources and fostering a more coordinated, beyond borders and polycentric approach to natural resource governance (Godar et al., 2016).

The discussion on the indirect anthropogenic drivers for changes in nature, nature´s contributions to people (NCP) and good quality of life is a relevant component of the Development Agenda 2030 and the Sustainable Development Goals (SDG). Equity, literacy level, share of population in extreme poverty, income distribution, access to public health, health care infrastructure, food security, political organization and socio-cultural aspects are relevant variables to define the critical mass of a country and the capacity of social debate, and hence its “anthropogenic assets”. On the other hand, the worldviews and culture (attitudes to environment/sustainability/equity), life-styles (including diets) and the level of societal tension and conflict are other important drivers of opposition or consensus in the economic and political arena. The level of efficiency in governance systems, the legislation and the strength of the institutions involved in decision-making and their implementation capacity, and their level of credibility and transparency, are also drivers that will influence the status and trend of NCP.

This section describes the current status and trends of six broad indirect anthropogenic drivers of changes in NCP in the Americas: Governance systems and institutions (4.3.1); Economic growth (4.3.2); International trade and finance (4.3.3); Population and demographic trends (4.3.4); Technological development (4.3.5); and Welfare and human development (4.3.6). Internationally comparable socioeconomic data for Greenland is limited in regional sources of the Americas, considering that Greenland has been politically and to some extent culturally associated with Europe for more than a millennium. Systematic socioeconomic data of other Protectorates located in the Americas were also not included in the following sections.

4.3.1 Governance systems and institutions (formal and informal)

There is a widespread consensus that governance (see definition Box 4.2) has a strong effect on environmental outcomes (Smith et al., 2003; Armitage et al., 2012; Delmas & Young, 2009; de Castro et al., 2016), although there is very limited empirical evidence relating governance measures to biodiversity and changes in ecosystem services.

In response to such consensus, there is a growing demand for governance arising from human-environment interactions, which nonetheless is escorted by a declining confidence in the capacity of governments to address such matters (Delmas & Young, 2009).

Rule of law, citizen’s rights of access to information, community participation and even access to justice have been recognized as a basis for poverty reduction and sustainable development as reflected by SDG16 “Peace, justice and strong institutions”. Evidence from the Americas reveals important differences across subregions for major governance indicators (defined Box 4.3) in the last two decades, as reported by the World Bank Figure 4.2.

Box 4.2. The meaning of governance

The broader definitions of governance are linked to international agencies (e.g. World Bank and Organization for Economic Cooperation and Development, OECD) and standards of "good" public governance (Armitage et al., 2012). These standards encompass accountability, transparency, responsiveness, equity and inclusion, effectiveness and efficiency, following the rule of law, and participatory, consensus-oriented decision making (Crabbé & LeRoy, 2008).

Environmental governance, as a subclass of the broader governance concept, has been defined as "the set of regulatory processes, mechanisms and organizations through which political actors influence environmental actions and outcomes" (Lemos & Agrawal, 2006), and it "should be understood broadly so as to include all institutional solutions for resolving conflicts over environmental resources" (Paavola, 2007).

Voice and accountability shows a decrease after 2004, except for the Caribbean islands. In turn, political stability and no violence fluctuated and decreased in North America until 2004 and then slightly recovered afterwards in all subregions. The other four indicators have remained largely stable over time according to public perception, with Mesoamerica and South America below the other two subregions. Yet, these aggregate figures hide particularities of specific countries and they should be taken carefully. These indicators have been criticized for their "construct validity", that is, whether the indicators measure what they intend to measure (Thomas, 2010), and for their methodology being too broad and biased (Langbein & Knack, 2010). These and previous critiques have been in turn contested (Kaufmann et al., 2007a, b), assuring the validity of the indicators and the methodological procedures.

Box 4.3. Definitions of governance indicators (Reproduced from Kaufmann et al. (2010)

Voice and accountability, capturing perceptions of the extent to which a country's citizens are able to participate in selecting their government, as well as freedom of expression, freedom of association, and a free media.

Political stability and absence of violence/terrorism, capturing perceptions of the likelihood that the government will be destabilized or overthrown by unconstitutional or violent means, including politically-motivated violence and terrorism.

Government effectiveness, capturing perceptions of the quality of public services, the quality of the civil service and the degree of its independence from political pressures, the quality of policy formulation and implementation, and the credibility of the government's commitment to such policies.

Regulatory quality, capturing perceptions of the ability of the government to formulate and implement sound policies and regulations that permit and promote private sector development.

Control of corruption, capturing perceptions of the extent to which public power is exercised for private gain, including both petty and grand forms of corruption, as well as "capture" of the state by elites and private interests.

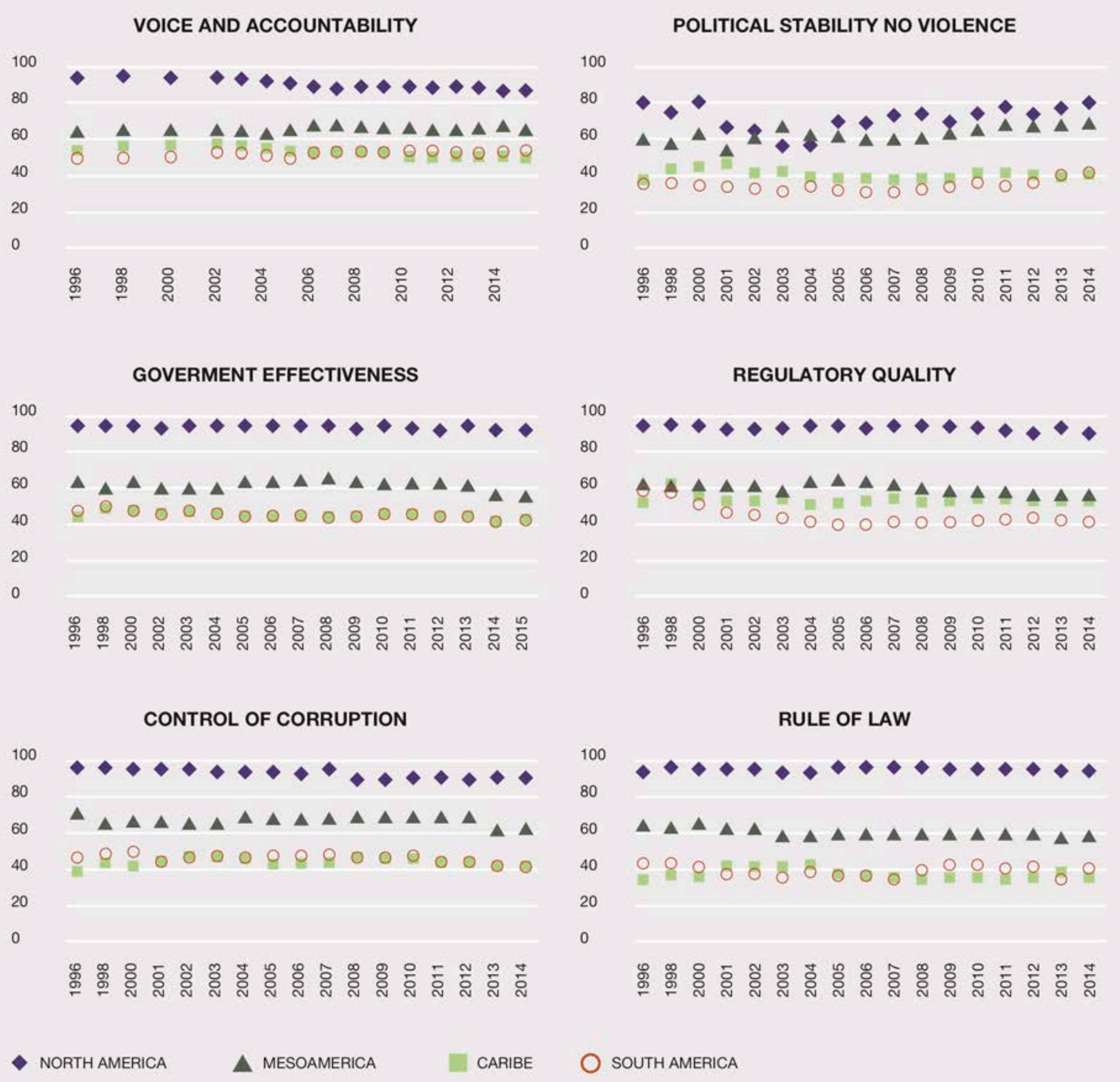
Rule of law, capturing perceptions of the extent to which agents have confidence in and abide by the rules of society, and in particular the quality of contract enforcement, property rights, the police, and the courts, as well as the likelihood of crime and violence (see Chapter 2, section 2.6).

Reinforcing the rule of law in the environmental domain from current levels is critical to the achievement of SDG and Aichi targets in the region. The importance of this matter was first recognized by the Rio Declaration and has been recently corroborated by the International Union for Conservation of Nature (IUCN) World Declaration on the Environmental Rule of Law in 2017. "Without the environmental rule of law and the enforcement of legal rights and obligations, environmental governance, conservation and protection may be arbitrary, subjective, and unpredictable" (IUCN, 2017).

On the other hand, the impacts of political instability on natural resources use have been tremendously negative in the region (Baud et al., 2011; Ruyle, 2017), particularly in South America in the last decade (Arsel et al., 2016). The most prominent conflicts concern mining in Brazil (see for example Tofóli et al., 2017), Ecuador (Avci & Fernández-Salvador, 2016), Honduras (Middeldorp et al., 2016) and Peru (Paredes, 2016), the use of rangelands for energy production (e.g. biofuels, solar) in the USA, Mexico and Canada (Kreuter et al., 2016), water use in most countries (Philpot et al., 2016), oil investments in Canada (Hebblewhite, 2017), and hydroelectricity projects on indigenous lands in Chile (Silva 2016), Colombia (Martínez & Castillo, 2016) and Canada.

Figure 4 2 Trends in World Bank Governance Indicators for the Americas between 1996 and 2014/2015, expressed in percentile rank, where lowest is 0 and highest is 100.

Source: own representation constructed from data available at http://info.worldbank.org/governance/wgi/#home.



Despite an impressive body of laws and institutions, the Region finds itself far off track in fulfilling the vision of sustainable development as indicated by the monitoring of the sustainable development goals (http://www.mdgmonitor.org). Political corruption (people exploiting public office for financial or other individual gain) is persistent in many countries and may have a significant impact on nature conservation by endorsing overexploitation of forests, wildlife, fisheries and other resources, and by impairing the effectiveness of conservation actions (Smith et al., 2003; Laurence, 2004). Few studies conducted in the region show the effect of corruption on biodiversity loss. Bulte et al. (2007) find a positive association between corruption and expansion of agricultural land (by subsidies), which is detrimental to forests in

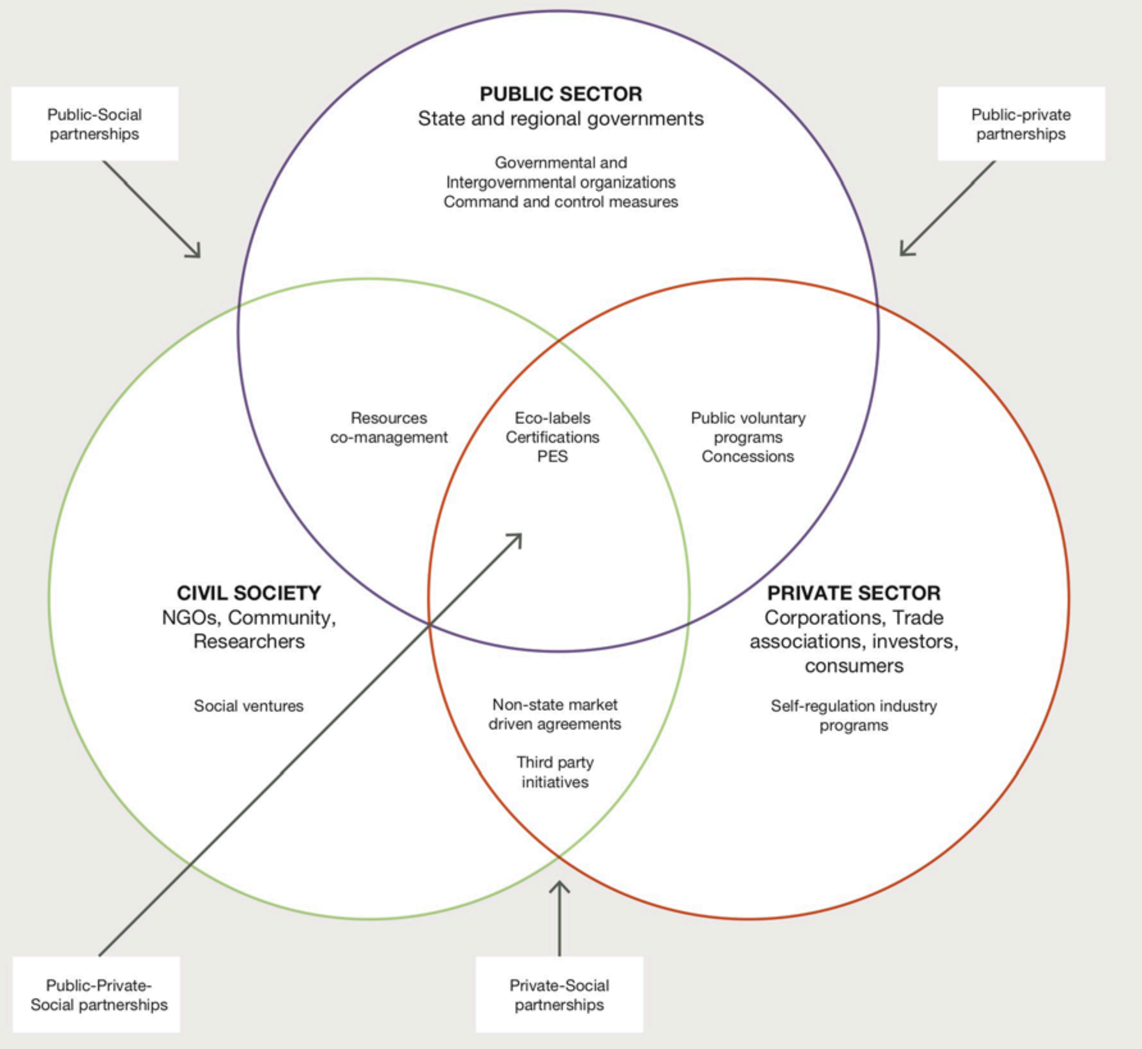
Latin America. Miller (2011) examines how corruption among forestry regulators in Costa Rica is one important factor that leads them to allow people to log illicitly. Yet, more robust studies showing causality between weak governance and biodiversity and ecosystem services loss are clearly needed for the Region.

Evolution of governance modes in the Americas and effects on nature conservation

Governments and States are no longer the most important basis of decision-making in the environmental field of the Americas. Instead, new actors (e.g. Non-Govermental Organizations (NGO)), researchers, indigenous groups) are performing critical roles and new mechanisms and forums are arising (e.g. The Economics on Economics and Biodiversity and IPBES) Figure 4.3 (Paavola, 2007; Armitage et al., 2012).

Different perceptions and values are strongly contested by different actors according to their images of nature (Sténs et al., 2016). Values, ideologies and sources of knowledge, which guide the manner in which nature is conceptualized, are key elements of environmental governance (de Castro et al., 2016; Inoue & Moreira, 2016) and they seem to be in increasing dispute. They influence how environmental issues are problematized, how solutions are planned, and how priorities and agreements are established between conflicting objectives. Therefore, the more actors involved in environmental governance, the more complex and heterogeneous the images become (de Castro et al., 2016; Tijoux, 2016).

Figure 4 3 Modes of environmental governance and actors involved. Source: Modified from Armitage *et al.* (2012).



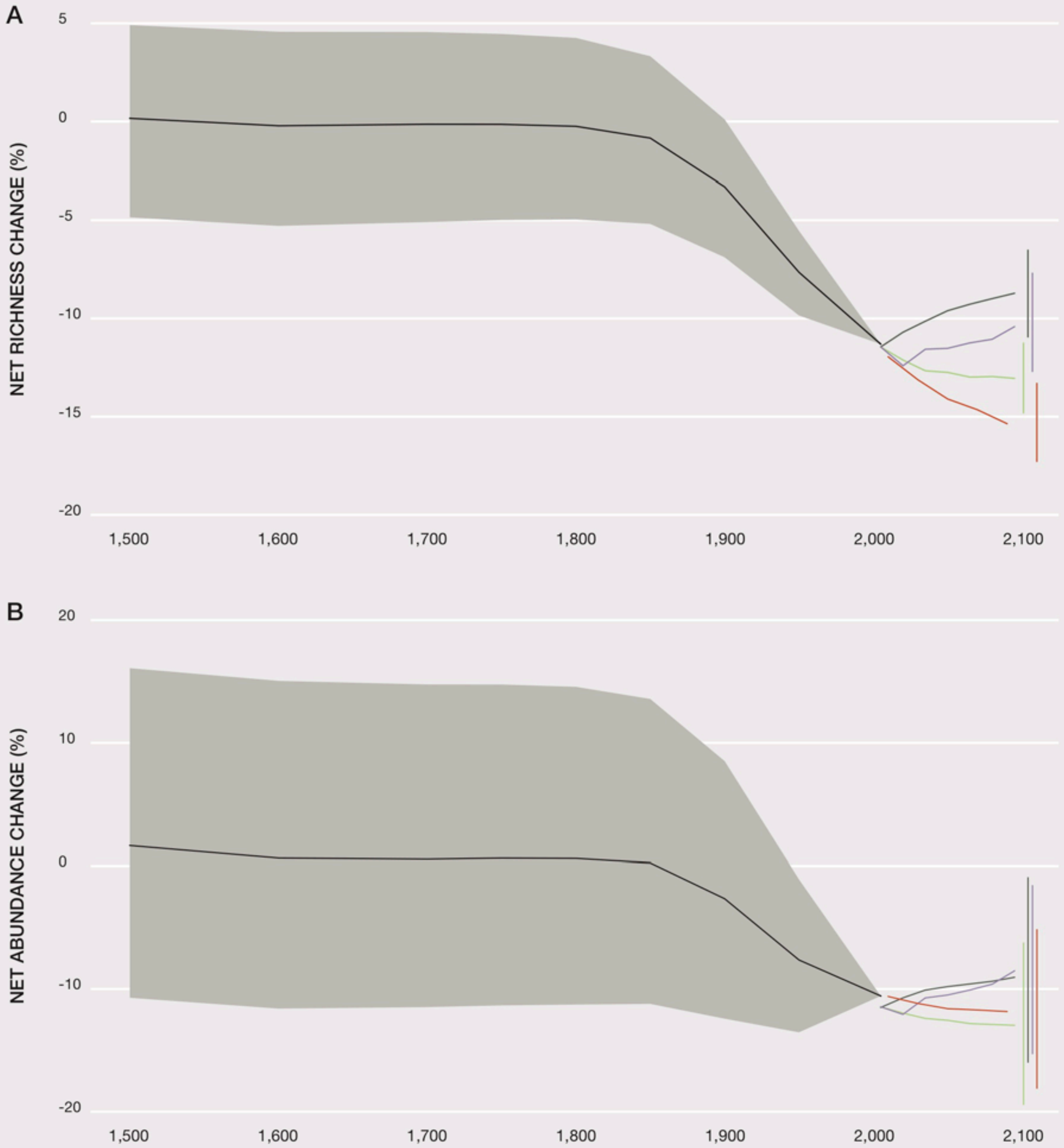
Environmental governance in the Americas has gone through major transformations in the last decades Figure 4.4 and yet biodiversity and ecosystem services continue to decline. From the mid-1980s onwards, most countries turned away from centralized, state-based institutional arrangements and direct regulation (Baud et al., 2011). Common problems around centralized modes of governance are the usual institutional fragmentation and centralization. A prominent example of these transformations is the case of the Great Lakes in the USA regarding water quality and water supply as key dimensions to be governed (Jetoo et al., 2015).

With the accent on privatization and decentralization, the new approaches towards management and conservation emphasized self-governance and higher levels of participation for civil society and private enterprises (Baud et al., 2011; de Castro et al., 2016).

Neoliberal policies guided the privatization of natural resources such as water (Molinos-Senate et al., 2015) and forests (Manuschevich, 2016) as in the case of Chile, and fish as in the case of the USA (Pinkerton & Davis, 2015; Carothers, 2015), along with land grabbing as in Argentina for example (Coscieme et al., 2016), producing major socio-environmental impacts (Liverman & Villas, 2006). In parallel, coalitions among civil society organizations, (international) NGOs and academic institutions established an alternative governance perspective for local communities, which was labeled participatory governance Figure 4.4. This new trend cemented the way for 'glocalization' processes linking local and global actors to develop local conservation approaches (Baud et al., 2011).

Figure 4 14 Historical and future estimates of net change in local diversity from 1500–2095, based on estimates of land-use, land-use intensity and human population density from the four RCP scenarios.

Net changes in richness A, total abundance B are shown. Historical (shading) and future (error bars) uncertainty shown as 95% confidence intervals. Source: based on Newbold *et al.* (2015).



By and large, the main governance arrangement towards nature conservation has been the centralized establishment of public protected areas (encompassing different levels of protection from total preservation to multiple uses). Comprising Mesoamerica, South America and the Caribbean the coverage of protected areas has increased by 8.9% with respect to the subregions' total area between 2000 and 2014, being the territory with the largest increase in area under protection worldwide (World Bank, 2017). The same three subregions show an increase between 2000 and 2014 of 5.2% of the total territorial waters protected with respect to the regions' total area. Conservation policy and implementation often assume that protected areas are enduring institutions, but some recent evidence suggests widespread protected areas downgrading, downsizing, and degazettement (Mascia et al., 2014). Mascia et al. (2014) describe protected areas downgrading, downsizing, and degazettement as a "patchy, episodic phenomenon" which nonetheless suggests tradeoffs between conservation goals and other policy objectives and is linked to industrial-scale natural resource extraction and development, local land pressures and land claims, and conservation planning.

Another circumstance is that in several cases the creation of protected areas has displaced local communities (Cardozo, 2011; Jones et al., 2017). For three case studies in Mexico, for example, García-Frapolli et al. (2009) identifies the most common difficulties in protected areas policy as: (1) uncoordinated public policies; (2) the usual conflict between environmental authorities and local people over the management of natural resources; and (3) the exclusion of local people's perspectives, values and beliefs in conservation policy development and implementation.

Aside from command and control arrangements such as protected areas, several hybrid modes have emerged in the region Figure 4.4. Among them the most notorious are: state private partnerships (certification), private-social partnerships (e.g. payment for ecosystem services), and co-management. forest certification is prominent in Brazil, Chile and Argentina (see Pinto and Mcdermontt, 2013; Cubbage et al., 2010). Another iconic example is the certification of coffee in countries such as Colombia, Brasil, Costa Rica, Ecuador, and Honduras, among others (Pinto et al., 2014; Ibañez and Blackman et al., 2016). Certification has recently expanded to industrial and smallscale fisheries with promising results in several countries of the region (Perez-Ramírez et al., 2016) (Box 4.4).

Box 4.4. The promise of fisheries certification

Ecolabelling and certification schemes are market-based tools to promote the sustainable use of natural resources. In the case of fisheries, ecolabels are a growing feature of international fish trade and marketing (Washington & Ababouch, 2011) in response to growing concerns about the state of the world's fish stocks, increased demand for fish and seafood, and a perception that many governments are failing to manage marine resources. The Marine Stewardship Council features as the most comprehensive fisheries certification scheme covering a range of species and dealing with all aspects of the management of a fishery. The Marine Stewardship Council has two standards: on sustainable fishing and on seafood traceability (Bush et al., 2013; Agnew et al., 2013; Washington & Ababouch, 2011). Although there are 10 Marine Stewardship Council-certified fisheries in Latin American and the Caribbean, this proportion is low (4%) compared to the total number of certified fisheries globally (Pérez Ramírez et al., 2016). Fisheries participating in the Marine Stewardship Council program in the region may be classified into two groups: (1) large enterprises of industrial fisheries, especially multi-national ones that can afford the certification process (i.e., Argentine hoki); and (2) small-scale fisheries that are vital to the local livelihoods (i.e. lobsters). Among the latter a successful case is the Chilean rock lobster (*Jasus frontalis*) of The Juan Fernández Archipelago and Robinson Crusoe Island, Marine Stewardship Council certified in 2015. The success of fishery management over recent years relies on five key management measures that are implemented with the full support of the community (near 900 inhabitants): only licensed artisanal fishers who are residents may harvest lobster in the area; the use of relatively small vessels that can only tend a few traps per day; informal property rights on individual fishing grounds; a conservative minimum landing size (115 mm length); and a closed season of four and a half months.

Despite the increasing enthusiasm for ecosystem services based market mechanisms, the reality is that incentive allocation on private lands has relied on scarce knowledge of ecosystem service supply by different properties (Ferraro et al., 2015). In the absence of supply data at the farm level for the entire region, the measurement of policy impacts has had to rely on imperfect proxies for additionality in terms of service provision (e.g. avoided deforestation) (Ferraro et al., 2015). Undeniably, the lack of complete, high-resolution, updated spatial information to obtain ecosystem services indicators is a primary restriction on the development of conservation planning assessments in developing countries (Di Minin & Toivonen, 2015; Stephenson et al., 2017) including the design of payment for ecosystem services mechanisms. In the domain of payments, Reduced Emissions from Deforestation and Forest Degradation and Reduced Emissions from Deforestation and Forest Degradation-Plus have emerged as a core climate change mitigation strategy. Nonetheless the mechanism has been harshly contested due to its undesirable social impacts and undetermined role in avoiding deforestation (Pirard & Belna, 2012).

The commitment by most countries to expand the area under protection in a representative and well-connected manner, as part of the Convention on Biological Diversity's (CBD) Aichi target 11, requires the inclusion of a range of protection mechanisms over a variety of tenures, including protected areas over private land (Woodley et al., 2012). Despite their potentially important role in biodiversity conservation, recognition of the role of private protected areas has suffered from sparse data, loose definitions and lack of integration within the broader conservation arena (Stolton et al., 2014) (see details in Box 4.5). The main challenges of private protected areas are the absence of recording and as a consequence there is no reliable information on how many there are, where they are located, what conservation activities they are engaged in. With private protected areas there is also an absence of clear guidelines for establishment and operation, and there are differences in the support and incentives given by government to the creation and maintenance of private protected areas (Bingham et al., 2017). They also face the challenge of avoiding conflicts with local and indigenous communities, particularly those located on the private protected areas' buffer zones (Serenari et al., 2017).

Box 4.5. The challenges of private protected areas

The declaration of private protected areas involves "a private intention to protect an area where government and other organizations do not play a pivotal role" (Stolton et al., 2014). The motivations behind their creation vary widely from pure philanthropic motives to real state and tourism development and speculation. The following are examples for different countries of the region (Stolton et al., 2014):

USA. There is no formal private protected areas definition and no comprehensive reporting, but there is an active private protected areas community driven by land trust organizations and NGOs, with many thousand private protected areas.

Canada. Private protected areas are primarily located on the country's southern border on land with high levels of species diversity and also species at risk.

Mexico. Private protected areas, which protect 487,300 hectares (0.25%) of the country's land surface, play an important role in connecting government managed protected areas.

Colombia. There are 280 registered national private protected areas organizations, most are small in area and many are in the Andes.

Chile. The term private protected areas is legally recognized, although undefined and unregulated. The private protected areas vary widely in size (from a few hectares to over 300,000 hectares) and ownership (comprising private individuals; industrial forest companies; NGOs; and foundations). They represent over 10% to the national protected area system.

Brazil. Brazil has a legislated and federated system of over 1,100 private reserves of natural heritage protecting approximately 703,700 ha.

At the local level, there hs been an emergence of community-based participatory conservation approaches seeking to engage local communities in management decisions, transfer rights to resources and allow sustainable use, to varying degrees. Many countries have introduced new policies and laws to support community-based conservation and there have been some successes (Box 4.6). However, in most cases, community-based conservation remains small-scale and isolated and is weakly integrated within the formal conservation sector (Baud et al., 2011; Lammers et al., 2017; Redmore et al., 2017) facing barriers such as a limited binding participation of communities in the development of conservation policies; insufficient devolution of authority and benefits to communities; and lack of support from other natural resource and economic sectors (Baud et al., 2011).

Box 4.6. Los pueblos del bosque

The socio-ecological struggles of traditional populations are what Martínez-Alier calls the “environmentalism of the poor” (Martínez-Alier, 2014). Within the multiple manifestations of this “ecology of the poor” in South America, Mesoamerica and the Caribbean, one of the first to have had an international echo was the movement of rubber tappers (*seringueiros*) who are not indigenous peoples but the first or second impoverished immigrants from northeastern Brazil, left in search of their own forms of subsistence long after the commercial exploitation of rubber on a large scale was over.

Acre rubber tappers formed unions, and in 1987 they joined the indigenous inhabitants of the Amazon to form an Alliance of Forest Peoples led by Francisco “Chico” Mendes who paid with his life for the cause of the Amazonian peoples (Tijoux, 2016). This movement was the forerunner of multiple expressions in the present as the Yasuní Park Project in Ecuador, which is considered one of the most important actions of the indigenous movements of the Americas. At present many of these actions are channeled through formal coalitions such as the Mesoamerican Alliance of Peoples and Forests http://www.alianzamesoamericana.org, among others.

On the opposite side of the green economy and the previous set of governance arrangements, new proposals arise that contemplate a fundamentally different ontology of nature, grouped under the label of *Buen Vivir* (Vanhulst & Bieling, 2014; Villalba-Eguiluz & Etxano, 2017) Figure 4.4. This trend includes a wide range of alternative conceptions of nature and of human-nature relations, starting with alternative, often indigenous, ideas about the relationship between human production, the environment and the rights of nature (Gudynas, 2011; Bauhardt, 2014). They propose a perspective of environmental governance that claims a transformation or even the end of the hegemonic capitalist model that is considered as the source of environmental degradation and injustice (de Castro et al., 2016; Inoue & Moreira, 2016).

These varied modes of governance do not necessarily coexist peacefully in the region and in many cases are antagonistic rather than synergistic, leading to severe social conflicts, which pose serious challenges for nature conservation and human well-being. Next to aspiration and creativity, attaining new modes to govern nature requires overcoming persistent barriers such as historical injustices, social inequalities and economic inefficiencies (Baud et al., 2011).

Major challenges have been reported in the past and continue to be significant limitations in the present. Among them: i) the environment continues to be a low priority (e.g. underfunded environmental agencies; low political support); ii) the understanding of environment-poverty-development links is frail (e.g. environmental concerns are perceived as barrier to economic growth); iii) the rule of law is weak (e.g. implementation of environmental legislation is still insufficient); and iv) environmental authority is weak (e.g. taking a management view rather than a governance focus). A critical issue pointed out at several international conservation forums is the fact that the three pillars of sustainable development – environmental, economic, and social – are not well integrated in the United Nations system and in global, regional, and national policies. Lessons learned in the past 25 years since the Earth Summit have led civil society organizations to uphold human rights as the basis for sustainable development governance.

4.3.2 Economic growth

Economic growth (measured as Gross Domestic Product (GDP) growth) is one of the main drivers of resource consumption (Dietz et al., 2007, quoted by IPBES, 2016). Virtually all socioeconomic and environmental scenarios for this century (i.e., up to the year 2050 and beyond) include economic growth as a key driver (IPBES, 2016).

Economic growth and trade can positively or negatively influence nature and NCP, but currently, on balance, they adversely impact nature and NCP when environmental and social development goals are insufficiently accounted for. Positive impacts of economic growth include, for instance, the resulting income availability for social and environmental investments, like biodiversity protection and conservation (Tlayie & Aryal, 2013), and greater environmental awareness. Negative impacts of economic growth mainly refer to the adverse consequences (e.g. habitat degradation, overharvesting, etc.) of those styles of economic growth that disregard social development and environmental goals.

Assessing relevant information on economic development includes consideration of key indicators, like regional and subregional GDP (and GDP per capita) growth trends; regional and subregional distribution of GDP purchasing power parity (PPP); as well as the sectoral structure of national economies (agriculture, industry, services). Table 4.1 synthetizes historical (since 1960) and projected (until 2050) trends for GDP and population in the Americas. GDP and population increased by 5.9 and 2.4 times, respectively, in the Americas from 1960 to 2016. By 2050, GDP in the Americas is expected to double with respect to 2016, while population would increase by 20% in that period.

Economic growth has been identified as a key driver of global greenhouse gasses emissions (IPCC, 2014a). With around 5% of world population, North America produces 24.2% of global GDP[11] (16.8% of global GDP_{PPP}) and 16% of global greenhouse gasses emissions, while Latin America and Caribbean accounts for 8.7% of total population, 7.6% of world GDP[12] (8.1% of global GDP_{PPP}), and 5.2% of global greenhouse gasses emissions (Table 4.1, IEA, 2016).

The impact of the consumers' purchasing power on the demand of natural resources is receiving growing attention in the economic literature nowadays due to the emergence of new waves of affluent consumers who tend to increase the demand for the limited natural resources (Myers & Kent, 2003). Purchasing power parity dollars are between 1.5 and 2.6 times higher than conventional dollars in at least 27 developing countries of the Americas. For the USA, PPP dollars and conventional dollars are the same by definition.

Table 4.1. Gross Domestic Product (GDP) and population in the Americas: historical and projected trends.

Regions	GDP PPP (*)	GDP *(**)*			Population		
	% of world GDP, 2016	% of world GDP, 2016	Cumulative change, 1960-2016 (GDP_{2016}/GDP_{1960})	Expected cumulative change, 2016-2050 (GDP_{2050}/GDP_{2016})	% of world population, 2017	Cumulative change, 1960-2017 (Pop_{2017}/Pop_{1960})	Expected cumulative change, 2017-2050 (Pop_{2050}/Pop_{2017})
North America	16.8	24.2	5.51	1.71	4.8	1.77	1.19
Mesoamerica	2.3	1.9	8.86	3.19	2.4	3.44	1.29
Caribbean	0.4	0.3	7.56	3.71	0.6	2.11	1.09
South America	5.4	5.4	7.05	3.16	5.7	2.86	1.18.
Americas	**24.9**	**31.8**	**5.86**	**1.98**	**13.5**	**2.37**	**1.20**

Notes:
(*) GDP at purchasing power parity (PPP)
(**) Constant 2010 USA Dollars
Reference data: World GDP_{PPP} in 2016: \$120.1 trillions; World GDP at Constant 2010 USA Dollar: \$75.5 trillions; World population 2017: 7,515.1 millions
Sources: Based on The World Bank Database (2017). https://data.worldbank.org/indicator/; Worldometers (2017). Accessed 2 May 2017, and 3 September 2017 at: http://www.worldometers.info/world-population/population-by-region/; Foure *et al.* (2012).

The countries of the region with the largest economies overall are the USA, Brazil, Canada, and Mexico. Dominica, Grenada, and Antigua and Barbuda, all small States in the Caribbean, have the region's smallest

[11] Based on constant 2010 USA Dollars (see Table 4.1)
[12] Based on constant 2010 USA Dollars (see Table 4.1)

economies overall. Factoring in countries' populations, the countries with the largest per capita incomes in the region are the USA and Canada. At around $50,000, their per capita incomes are considerably higher than all other countries in the region. The other countries' per capita incomes vary between Haiti's low of about $728 to The Bahamas $19,758. In general, per capita incomes are lowest in the Mesoamerica subregion, though other subregions exhibit a fair degree of variation (World Bank, 2017[13]).

The economies of the Americas vary widely in the sectoral composition of their national output. The contribution of agricultural production to national output has fallen to less than 20% throughout the region with the exception of Haiti where agriculture's GDP share is 21.5%. The economies of the region are primarily service driven, although there is variation across the individual national economies between Paraguay's 51.2% to Barbados' 85.5%. Throughout the region, the countries with the higher per capita incomes are those whose economic output is driven more heavily by service sector activity. Industrial production is a significant driver of most of the economies of the region, ranging from contributing less than 15% of GDP in Barbados and Grenada, to more than 40% of Trinidad and Tobago's GDP. Most economies in the region derive 25% to 35% of their GDP from industrial production (World Bank, 2017[14]).

The Americas has experienced substantial economic growth since 1960. Although the worldwide recession of 2008-2009 temporarily reduced national incomes, GDP in the Americas has increased approximately six fold since 1960, although North American income grew from a substantially higher 1960 level. Despite increasing populations throughout the region, the pace of real GDP growth has been sufficient to raise per capita GDP more than twofold from 1960 to 2015 (World Bank, 2017[15]).

While overall growth has been sizable at the regional and subregional levels, individual countries within the Americas have experienced varying growth trends since 1960. Per capita incomes have increased substantially over time in some countries; in other countries, per capita incomes have increased more modestly, or in still other countries, barely at all. In North America, Canada and the USA each experienced large growth in per capita income from already high 1960 levels. In Mesoamerica, GDP per capita grew significantly in Panama, Costa Rica, and Mexico, while it increased much more slowly in other countries. In the Caribbean subregion, The Bahamas has consistently had significantly higher per capita income than the rest of the subregion, followed by Trinidad and Tobago, Barbados, St. Kitts and Nevis, and Antigua and Barbuda. Incomes in a handful of the subregion's countries barely grew at all. In South America, per capita GDP shows varying growth by country. Venezuela's was higher on average (partly due to oil endowments). In 1960, per capita incomes in the subregion (excluding Venezuela) ranged from about $1,000 in Paraguay to about $5,600 in Argentina. By 2015, the range had widened considerably, from about $2,400 in Bolivia to almost $15,000 in Chile. Countries with strong growth since about 1990 include Chile, Uruguay, Brazil, Argentina, and Suriname. Peru's growth has been steady but slower, with a recent acceleration (World Bank, 2017[16]).

The GDP growth rate for the USA fell from an average of 3.3% per year in 1997-2006 to 1.2% in 2007-2015; while the economic dynamics for Latin America and the Caribbean also diminished from an average of 3.1% to 2.9% in those years. These trends partially reflect the interconnections between the USA market and Latin America and the Caribbean economies, particularly those of Mesoamerica and the Caribbean. The period 2007-2015 was characterized by the effects of the global economic crisis, with absolute reductions of GDP for the USA in 2008 (by -0.3%) and in 2009 (by -2.8%), and for Latin America and the Caribbean region in 2009 (by -1.2%) and 2015 (by -0.3%)[17].

Growing pressures on natural resources are expressed in different ways in different country groupings and regions, due to patterns indicating high per capita consumption of natural resources, growing dependency on commodities exports and other conditions (Table 4.2). Per capita consumption of natural resources is particularly high in North America. For instance, total primary energy consumption per capita for North

[13] Data available at http://databank.worldbank.org/data/reports.aspx

[14] Data available at http://databank.worldbank.org/data/reports.aspx

[15] Data available at http://data.worldbank.org/indicator/

[16] Data available at http://data.worldbank.org/indicator/

[17] Based on IMF (2014 & 2015).

America was 6.1 tons oil equivalent versus 2.39 tons oil equivalent for non-OECD Americas in 2013 (IEA, 2015; Pichs, 2008). According to WWF (2014) the nitrogen loss indicator[18] is largest in North America (81kg/capita/year), more than twice the world average (29 kg/capita/year).

Commodities (including, for instance, hydrocarbons, mineral raw materials, food and other agricultural products) represent more than 50% of Latin America and the Caribbean exports (for the years 2012-2014) and 9% of the regional GDP, reflecting a clear extractivist bias in the regional economic growth. South America is the most commodity-intensive subregion in the Americas, with commodities accounting for more than 70% of goods exports, and nearly 10% of GDP. Mesoamerica is considerably less commodity dependent than South America, but commodities still account for about one quarter of exports there, and 7.5% of GDP (World Bank, 2016). North American economies are more diversified than Latin America and the Caribbean economies and consequently less vulnerable to price shocks in the global commodity markets. The export diversification index[19] for North America in 2015 was 0.213, while this indicator averaged 0.584 for the Caribbean, 0.549 for South America, and 0.375 in Mesoamerica (UNCTAD, 2016).

Table 4.2. Combining GDP growth and GDP intensity in natural resources (including energy / carbon intensity) and assessing the level of pressure on biodiversity and ecosystem services.

Table 4 2 Combining GDP growth and GDP intensity in natural resources (including energy / carbon intensity) and assessing the level of pressure on biodiversity and ecosystem services

Note: GDP intensity in natural resources refers to the consumption of natural resources required to produce a unit of GDP, with fossil fuel intensity (measured as volume of oil equivalent for monetary unit of GDP), for instance, being a subset of GDP intensity in natural resources. Source: Elaborated by the authors based on ECLAC (2014); CEPAL (2017); IMF (2017); The World Bank Database (2017); UNCTAD (2016); WWF (2014, 2016); GFN (2017).

Low GDP growth / High GDP intensity in natural resources High pressures, due to situations like economic stagnation (e.g. Extractivist policies with economic crisis)	**Low GDP growth / High GDP intensity in natural resources** Very high pressures, through very high GHG emissions (reinforcing climate change), land use change (deforestation) and general overexploitation of natural resources (e.g. Extractivist policies with economic expansion).
Low GDP growth / High GDP intensity in natural resources Low pressures associated, for instance, to low technological development.	**Low GDP growth / High GDP intensity in natural resources** Low pressures due to de-coupling between GDP growth and GDP intensity.

Rapid economic growth generates growing pressures on nature and NCP, particularly when the economic growth is heavily dependent on increasing use of natural resources and carbon intensity. Economic crisis also increases pressures on natural resources when economic agents tend to compensate low commodity prices with higher export volumes.

In recent decades, the increase in household income in Latin America and the Caribbean has resulted in a striking rise in consumption. Per capita private consumption for Latin America and the Caribbean, in USA dollars at constant 2010 prices, rose by a cumulative annual rate of 2% between 1990 and 2000 and 2.5% between 2000 and 2016, while the corresponding rates for North America were 2.5% and 1.1%, respectively in those periods. Since 2010, the average per capita private consumption in Latin America and the Caribbean have surpassed the world average, but by 2016 it was only 17.1% of the corresponding level for North America (Table 4. 3).

[18]The nitrogen loss indicator was developed for the CBD and represents the potential nitrogen pollution from all sources within a country or region as a result of the production and consumption of food and the use of energy (WWF, 2014).

[19] The export diversification index is calculated by measuring the absolute deviation of the export structure of a country from world structure. This index takes values between 0 and 1. A value closer to 1 indicates greater divergence from the world pattern (UNCTAD, 2016)

Table 4.3. Household final consumption expenditure per capita.

Regions	Constant 2010 USA Dollars			Average annual % growth	
	1990	2000	2016	1990-2000	2000-2016
North America	22675	28703	34841	2.5	1.1
Latin America and the Caribbean	3675	4488	5958	2.0	2.5
World	4036	4710	5833	1.6	1.4

Source: The World Bank Database (2017) World Development Indicators. https://data.worldbank.org/indicator/NE.CON.PRVT.PC.KD?view=chart. Accessed 4 November 2017

Within Latin America, consumption trends have followed differentiated patterns across the various subregions in the last three decades. The expansion of private consumption in South America, for instance, has been supported, to a large extent, by the boom in exports of renewable and non-renewable natural resources, with highly favorable terms of trade up to 2014. In Central America, however, the consumption dynamics have been more closely associated with the stabilization of remittances, while Mexico combines both patterns: exports of natural resources (mainly oil) and significant flows of remittances (ECLAC, 2014). The prevailing consumption model in Latin America and the Caribbean is still what Fernando Fajnzylber termed *"showcase modernization"*, which may expand the population's access to goods and services but also tends to replicate the socio-environmentally unsustainable conditions seen in the developed countries (ECLAC, 2014).

On the one hand, private consumption dynamics in Latin America and the Caribbean during the recent decades has brought positive effects, as it has been partially associated with increased well-being in sectors that were deprived in the past, and it has contributed to better living standards, which in turn enable better use of time and more opportunities for capacity-building. On the other hand, growing private consumption has also brought negative consequences and externalities such as higher fossil fuel consumption, waste generation, air pollution, environmental destruction and increased exploitation of renewable and non-renewable natural resources. In addition to that, consumption in Latin America and the Caribbean is pro-cyclical and exposes economies to greater vulnerability. Recent regional consumption trends have also widened the gap between consumers of private and public services (ECLAC, 2014).

Another source of concern is that the upper income segments of the population in Latin America and the Caribbean, favoured by wealth concentration, tend to show a pattern of consumption very intensive in high-cost private services and luxury goods, with a high imported content. The region's highest income quintile spends between four and 12 times more than the lowest income quintile (ECLAC, 2014).

Scenarios that assume rapid economic growth in the coming decades are mainly based on prioritizing market goals and incentives under conventional market approaches, with adverse social and environmental implications, including negative impacts on biodiversity and ecosystems (e.g. Global Environmental Outlook 4 Market First (IPBES, 2016)).

Statistics on the composition of the ecological footprint for the Americas reveal that the carbon footprint accounts for 53% of the total ecological footprint of the Western Hemisphere (65% for North America). The second largest hemispheric contributor is cropland, which accounts for 19% (26% in South America), and the third position is shared by grazing land and forest products (12% each). The predominant role of the carbon footprint in the Americas is mainly associated with the high dependency on fossil fuels in the region (¡Error! No se encuentra el origen de la referencia.).

The list of the top five countries with the highest ecological footprint includes two countries from the Americas, the USA (accounting for 13.7% of world total ecological footprint) and Brazil (with 3.7%) (WWF, 2014).

Table 4.4. Composition of the ecological footprint the regions of the Americas (%).

Regions	Cropland	Grazing land	Forest products	Fishing Grounds	Building-up land	Carbon	Total ecological footprint
North America	16	5	11	2	1	65	100
Mesoamerica (1)	22	12	12	2	2	50	100
Caribbean (2)	25	10	7	4	2	52	100
South America (3)	26	30	16	1	4	23	100
Latin America & Caribbean	25	25	14	1	4	31	100
Americas	**19**	**12**	**12**	**2**	**2**	**53**	**100**

Notes: Ecological footprint data for 2013. Composition in % of the ecological footprint for 2010. 1. Information for Belize is not available, 2. Information available only for five countries: Cuba, Dominican Republic, Haiti, Jamaica and Trinidad and Tobago, 3. Information for Guyana and Suriname is not available. Source: Based on WWF (2014, 2016), GFN (2017) (See Chapter 2, section 2.6).

4.3.3 International trade and finances

Economic activities, international trade and financial flows are closely related, particularly in recent decades due to the expansion of economic globalization. Trends in economic growth, international trade and financial markets considerably influence changes nitrogen, NCP and good quality of life through various direct and indirect pathways. In turn, these pathways are influenced by a number of policy channels and mechanisms, like trade policies, including incentives (tax exemptions, subsidies) and trade barriers, the dynamics of foreign debt and foreign debt service, flows of foreign direct investments, and monetary policies (dynamic of exchange rates, interest rates).

The Americas generates around 18% of world exports, and most of this proportion (12.6%) is supplied by North America. Latin America and the Caribbean contribution to world exports (5.4%) is modest in relation to the region´s fraction of world population (8.7%).

The volumes of trade are directly related to economic size and openness. The USA has the highest trade volumes, with a substantial trade deficit. Canada and Mexico are in the next tier with respect to volumes, followed by Brazil. The composition of trade reflects countries' economic activity and natural resources. Fuel ranges between 10% and 23% of imports for all countries in the region except Costa Rica and fuel exporting countries. Over 50% of all countries' goods imports are manufactured goods. Manufactured goods form over ¾ of all imported goods for 11 of the countries with data. On the export side, agricultural raw material forms a very small part of each nation's trade. It is most important for Uruguay, comprising 12.7% of its merchandise exports. Fuel comprises over half of Venezuela's, Colombia's and Bolivia's exports and plays an important role in exports from Ecuador and Canada. Manufactured goods form an important component of most of the region's nations' exports, being most important for Mexico and El Salvador. Tourism is by far the most important export for The Bahamas, and is also important to other Caribbean nations (World Bank, 2017[20]).

As mentioned before, natural resources (oil, minerals, and agricultural products) contribute with more than 50% to Latin America and the Caribbean exports. Commodities account for more than 70% of exports in South America, and about one quarter of exports in Mesoamerica (World Bank, 2016). Tourism is also a key sector in several Latin America and the Caribbean countries, particularly for small Caribbean island States and some Central American countries. Drastic reduction of commodities prices in world markets since 2014

[20] Data available at http://data.worldbank.org/indicator/

has severely affected commodities exporters in the region. In some cases, Latin America and the Caribbean countries have tried to compensate declining export prices of commodities with increasing export volumes, generating additional pressure on the natural environment. International export prices for Latin America (19 countries reported by ECLAC) declined by 8.7% in 2015 in relation to 2010, while export volume increased by 15.4% (CEPAL, 2015). As indicated before, the export structure of North America is more diversified, and therefore these developed economies are less vulnerable to market shocks, in relation to the Latin America and the Caribbean economies.

In contrast to North American economies, most Latin America and the Caribbean countries have very limited influence in world trade and financial markets and flows, with high vulnerability to abrupt changes in those markets (Table 4.5)

Table 4.5. Relevant trade data for the Americas (2016).

Country /Region	Number of economies	% of world exports of goods and services	Exports of goods and services as % of GDP
North America	2	12.6	14.0
Mesoamerica	8	2.2	37.0
Caribbean	13	0.4	22.3
South America	12	2.8	16.5
Latin America & Caribbean	32	5.4	21.7
Americas	34	18.0	15.6

Source: The World Bank (2017). World Development Indicators (Last Updated Date: 08.02.2017): www.worldbank.org

The Table 4.6 presents the potential pressures on nature and NCP due to the dynamics of trade and financial trends. In South America, for instance, export policies and currency exchange rates (Richards et al., 2012) have created incentives to buy land for planting soybean, and this explains the high deforestation rate in ecosystems like the South American Chaco. This has generated not only high export revenues but also the devastation of nature as well as increasing poverty and social conflicts (Barbarán, 2015; Barbarán et al., 2015; Weinhold et al., 2013).

The cumulative foreign debt for Latin America and the Caribbean countries reached $2,062 billion in 2016, with a per capita foreign debt for the region of $3,250. Total cumulative payments of foreign debt service (interests and amortization) increased to $3,461 billion during 2008-2016. The regional payments to cover the foreign debt service accounted for 51.4% of Latin America and the Caribbean export income (including goods and services) in 2016 (based on IMF, 2014, 2015, 2016, 2017). South America absorbs 70% of regional Latin America and the Caribbean foreign debt (corresponding 22% to Brazil); Mesoamerica, 27% (with Mexico absorbing 21%); and the Caribbean, 3% (based on CEPAL, 2016).

Foreign debt for North America reached around $20.6 trillion in 2016 / early 2017 (corresponding 89% of this amount to the USA[21]). Approximately 80% of USA foreign debt is denominated in USA dollars. Foreign lenders have been willing to hold USA dollar denominated debt instruments because they perceive the dollar as the world's reserve currency. With the USA dollar being the national currency of the USA, this makes a significant qualitative difference between the foreign debt status of North America with regard to other regions of the Americas.

The flow of foreign direct investments to the Latin America and the Caribbean region totaled $134.8 billion in 2015 (8% below the average flow for the period 2011- 2014). This trend has been influenced to a great extent by the declining tendency of prices for commodities exported by the region. South America hosted 73% of foreign direct investments flows to Latin America and the Caribbean in 2015 (only Brazil, 46%);

[21] http://ticdata.treasury.gov/Publish/debta2017q1.html; http://www.indexmundi.com/united_states/debt_external.html; http://www.statcan.gc.ca/tables-tableaux/sum-som/l01/cst01/indi01j-eng.htm.

Mesoamerica, 24% (only Mexico 16%) and the Caribbean, 3% (based on CEPAL, 2016). Foreign direct investments inflows to North America reached $428.5 billion in 2015 (only USA 89%) (UNCTAD, 2016).

Table 4.6. Potential pressures on biodiversity and ecosystem services due to the dynamics of trade and financial trends.

Trade & Finance Indicators	Case 1	Case 2
Prices for relevant export products based on natural resources (including carbon intensive exports).	**High prices:** Potential pressures on biodiversity due to the incentive of having high export prices. New exporters can emerge.	**Low prices:** Potential pressures due to attempts to compensate losses in export prices with increasing export physical volumes.
Trade Policies for trading products based on natural resources.	**Restrictive policies (e.g. protectionist measures / trade barriers):** Potential pressures on biodiversity in the importing countries as non-efficient producers may be competitive. Growing pressures on biodiversity in exporting countries, due to efforts to find alternative export solutions with limited options.	**Non-restrictive polices (e.g. trade liberalization):** Significant pressures on biodiversity when these measures are not carried out in a sustainable development context, as they may encourage a massive flow of trade.
Foreign Debt (in proportion to key indicators like GDP and/or export. income).	**High levels**: Significant pressures on biodiversity in debtor countries, as they struggle for get additional income to serve the foreign debt, with one option being increasing export of products / services based on natural resources.	**Low levels :** Low pressure on biodiversity.
Foreign Direct Investments (particularly in sectors based on natural resources).	**Growing flows:** Significant pressures on biodiversity in the recipient country, particularly in absence of well- established local foreign direct investments laws to ensure sustainable use of natural resources.	**Declining flows:** Pressures on biodiversity would depend on local investment options as alternative to foreign direct investments.
Monetary Policies.	**E.g. Local currency devaluation:** This encourages exports, by making them more competitive. This could imply additional pressures on biodiversity.	**E.g. Local currency revaluation:** This makes exports less competitive. This could imply pressures on biodiversity in exporting countries, due to efforts to find alternative export solutions with limited options.

Note: Cases (1 and 2) correspond to each indicator of the first column (horizontal analysis).
Source: Elaborated by the authors. Based on ECLAC (2014), CEPAL (2017), IMF (2017), The World Bank Database (2017), UNCTAD (2016), WWF (2014, 2016).

4.3.4 Technological development

Human development has been historically related to technological change, with historical epochs named after the key technologies: the Stone, Bronze and Iron Ages, the industrial revolution, the age of steam, and the information age. The way of orienting the development, dissemination, and use of technology is crucial to find just, equitable, and sustainable solutions for present and future generations. Political, social, cultural, and economic factors determine the way new technologies are developed and used (Trace, 2016).

The rate of technological change is considered as an indirect driver of changes in nature, NCP and good quality of life because it affects the efficiency by which ecosystem services are produced or used (Alcamo et al., 2005, quoted by IPBES, 2016). The impact of technological innovation on biodiversity and ecosystem change is exerted through its influence on direct drivers (e.g. land use change), as well as through interactions and synergies with other indirect drivers (e.g. economic growth, see 4.3.2).

Finding indicators of the status and trends in the Americas region's or any given country's technological development is difficult due to data shortcomings. The Americas, with 13.6% of world population (2013 data) accounted for 22.5% of the total amount of researchers, 33.1% of world investments in research and development, 34.8% of world publications and 53.2% of patents submitted to the US Patent and Trademark Office. Regional information reveals the persisting gaps regarding science, technology and innovation in the Americas (¡Error! No se encuentra el origen de la referencia.).

Table 4.7. Selected science and technology indicators in the Americas (2013)[1].

Countries	% of world population, 2013	% of world R&D, 2013	Per capita R&D (USD), 2013	R&D/ GDP, 2013 (%)	Researchers / thousand inhabitants, 2013	% total researchers, 2013	% of global increase in R&D 2007-2013	% of world publications, 2014	% of total patents submitted to USPTO, 2013 [4]
USA	4.3	28.1	1249.3	2.81	4.0	16.7	10.8	25.3	50.1[5]
Canada	0.5	1.5	612.0	1.63	4.5	2.1	[2]	4.3	2.8
Latina America	8.1	3.4	87.2	0.69	0.5	3.6	4.2	5.1	0.3
Caribbean	0.6	0.1	40.8	0.34	0.2	0.1	0.0[3]	0.1	0.0
World	**100**	**100**	**206.3**	**1.70**	**1.1**	**100**	**100**	**100**	**100**

Notes:
[1] This information does not separate non-military and military research and development (R&D).
[2] Canadian investments in R&D reduced from $23.3 billion in 2007 to $21.5 billion in 2013.
[3] Caribbean investments in R&D marginally increased from $1.6 billion in 2007 to $1.7 billion in 2013.
[4] UPSTO: United States Patent and Trademark Office.
[5] This is used as an international indicator considering the attractiveness of the USA market also for foreign investors.
Source: UNESCO (2016).

Most of the scientific and technological potential of the Americas corresponds to North America, with 18.8% of researchers, 29.6% of global research and development, 29.6% of world publications, and 52.9% of patents submitted to US Patent and Trademark Office. Latin America and the Caribbean only account for 3.7% of researchers, 3.5% of global research and development, 5.2% of publications and 0.3% of US Patent and Trademark Office patents. The USA accounted for 10.8% of the global increase of research and development during 2007-2013, while the contribution to that increase from Latin America and the Caribbean hardly reached 4.2% Table 4.7**,** UNESCO, 2016).

The availability of secure internet servers in the Americas has increased rapidly since the early 2000s. The North American subregion significantly outpaces the Latin America and the Caribbean subregion, however. In North America, there are currently almost 1,600 servers per million people, while in Latin America and the Caribbean there are only 59 per million people. Individual countries within subregions also exhibit wide variation in both the current number and increase in the number of secure internet servers per million people (World Bank, 2017[22]).

Technological innovation can catalyze paradigm shifts in production systems (Pérez, 2004, quoted by IPBES, 2016) that cause biodiversity loss and adverse ecosystem changes (i.e. technologies as part of the problem), or conversely reduce biodiversity loss and improve ecosystems health (technologies as part of the solution).

[22] Data available at http://data.worldbank.org/indicator/

Technology offers important positive solutions to resource conservation, sustainable use and development, and management, but technological change can also increase pressure on ecosystem services through increasing resource demand and leading to unforeseen ecological risks, particularly for technologies associated with agriculture and other land uses (e.g. first generation of biofuels when produced unsustainably).

As part of the solution space, technological change can increase agriculture efficiency and replace unsustainable production patterns (e.g. improvements in crop yields and resilience, sustainable livestock, fishing, and aquaculture practices). Although technology can significantly increase the availability of some ecosystem services, and improve the efficiency of provision, management, and allocation of different ecosystem services, it cannot serve as a substitute for all ecosystem services (Carpenter et al., 2006, quoted by IPBES, 2016).

In some cases, technological developments and agricultural practices may combine positive and negative implications for biodiversity and ecosystems as revealed by the agricultural intensification of the “green revolution”. On the one hand the “green revolution” led to higher crop yields and lower food prices, partially mitigating the expansion of agricultural land and resulting in a net decrease of greenhouse gasses emissions. On the other hand, excessive nitrogen and phosphorous use through fertilizers, associated with the “green revolution” led to substantial degradation of freshwater and marine habitats. In addition, the shift from traditional crop varieties to industrial monocultures resulted in a loss of crop genetic diversity as well as increased susceptibility to disease and pests (IPBES, 2016, chapter 3). This confirms the importance of promoting sustainable practices with an integrative approach concerning the linkages between environment and socioeconomic development.

Those production technologies and practices that are based on increasing dependence on external inputs like chemical fertilizers, pesticides, herbicides and water for crop production and artificial feeds, supplements and antibiotics for livestock and aquaculture production have adverse implications in terms of sustainability. These technologies damage the environment, undermine the nutritional and health value of foods, lead to reduced function of essential ecosystem services and result in the loss of biodiversity (FAO, 2011, quoted by Trace, 2016).

When the technological changes in agriculture are implemented in accordance with the principles of sustainable development, these transformations may imply greater equity within and between generations, including with regard to food security (FAO, 1996).

Agroecological food production systems are considered as one approach to addressing the loss of biodiversity and the consequent unsustainability of industrialized food production, because they recognize the interdependencies between the sources of food and the wider environment, and the overlapping needs to provide sustainable food systems and sustainable livelihoods (Trace, 2016). Local knowledge and culture can be considered as integral parts of agricultural biodiversity (FAO, 2004, quoted by Trace, 2016). Agroecology considers productive processes in a broad and integral manner, taking into account the complexity of local forms of production. It is based on sustainability criteria, resource conservation and social equity (Vos et al., 2015).

The misappropriation of traditional biodiversity knowledge or ‘biopiracy’ has been considered as one of the most ‘complex problems facing the future of traditional knowledge’ (Khor, 2002, quoted by Trace, 2016). The system of community sharing and collaborative innovation is being challenged by intellectual property rights and the trade-related aspects of intellectual property rights regime, which together create a new system to exert private ownership rights over knowledge (Trace, 2016).

The intersection between agriculture, trade, and intellectual property governance is marked by a diversity of institutions involved, including the World Trade Organization, the World Intellectual Property Organization, the CBD, and the Food and Agriculture Organisation. On balance, the corporations have the upper hand in this complicated game (Sell, 2009).

A combination of expanded intellectual property rights and relaxed antitrust enforcement facilitated a recent shift from public to private provision of seeds, which is undermining small farmers’ tradition of saving seeds and reusing seeds. In this and other ways, the current situation is marked by underinvestment in crops and technologies suitable for smallholder farmers. In agri-biotechnology, six companies alone hold 75% of all USA patents granted to the top thirty patent-holding firms (Dutfield, 2003; Fowler, 1994). The top ten seed companies control over half of the global seed market (ETC Group, 2008) and are contributing to monoculture and associated loss of biodiversity in Latin America. This institutional dominance of transnational corporation facilitates "gene grab" (Sell, 2009), with negative effects on biodiversity, competition, and food security to the extent that it prevents resource sharing and locks out potential user-innovators by preventing small farmers from breeding, saving and reusing seeds to feed themselves and their communities (Rajotte, 2008). This is especially consequential considering that small farmers provide the majority of the food consumed by national populations. In Brazil, small farmers occupy 30% of agricultural land yet produce 70% of the food consumed by Brazilians.

4.3.5 Population and demographic trends

Assessing human demographic trends and their implications for nature, NCP and good quality of life includes consideration of total population and age structure; urban vs. rural populations and urban forms; information on locations, like coastal versus inland, migration flows, among other indicators Table 4.8 and present data on population and demographic trends in the Americas for the period 1960-2017 and expected future trends to 2050.

The Americas accounted for 13.5% of the world's estimated population in 2017. Subregionally, while having nearly equal areas[23], North America accounts for 4.8% of world population, while Latin America and the Caribbean accounts for nearly twice that at 8.7% of world population. This is reflected in population density, with Latin America and the Caribbean being much more densely settled (32 people per km^2) than Northern America (20 people per km^2). The population of the Americas is highly urbanized, with 80.8% of the region's population residing in urban settings (82.8% for North America, and 79.7% for Latin America and the Caribbean) Table 4.8 (Index Mundi, 2017).

Urbanization, driven by growing populations and internal migration, acts as an indirect driver of land-use change through linear infrastructures like transportation networks, as well as through synergies with other forms of infrastructure development (Seiler, 2001, quoted by IPBES, 2016, see also section 4.4.1). In Latin America and Caribbean 35% of the population (year-basis 2015) gained access to sanitation since 1990, but still 12% of the urban population and 36% of rural population do not have access to improved sanitation facilities (UN-Habitat, 2016). On average, only 50% of the population in Latin America is connected to sewerage and 30% of those households receive any treatment. The poor systematic waste management in Latin America and the Caribbean implies in pollution of inland waters and coastal areas (4.4.2), affecting biodiversity and human health.

[23] Area data are not corrected for inhabitable spaces.

Table 4.8. Population in the Americas by region in 2017.

Regions	Population 2017	Yearly Change, %	Migrants (net)	Median Age	Fertility Rate	Density (P/Km²)
North America	363,224,006	0.75	1,219,564	38.4	1.86	20
Mesoamerica	177,249,493	1.28	-192,495	26.9	2.34	72
Caribbean	43,767,545	0.64	-120,068	30.5	2.27	194
South America	426,548,298	0.95	-63,786	30.6	2.03	24
Americas	**1,010,789,342**					

Source: Authors' compilation from Worldometers (2017). Accessed 2 May 2017, and 3 September 2017 at http://www.worldometers.info/world-population/population-by-region/

Current population growth rates are 0.75% per year in North America and 1.02% per year in Latin America and the Caribbean. Migration and fertility rates combine differently in these two subregions. In Latin America, an above-replacement fertility rate of 2.15 outweighs net outmigration from the subregion, such that population growth is positive and relatively high compared to the world community there. In the North American subregion, net in-migration outweighs lower-than-replacement fertility rate to produce that subregion's positive population growth rate. North America has among the world's oldest median population, while Latin America and the Caribbean has among the world's youngest median population.

Table 4.9. Population in the Americas by region: present (2017), past (1960-2017) and future (2017-2050) trends.

Regions	Region's share of world pop, 2017	Region's share of Americas pop, 2017	Total pop change, 1960-2017 (Pop_{2017} / Pop_{1960})	Total pop change, 2017-2050 (Pop_{2050} / Pop_{2017})	Urban, % of total pop 2017	Urban pop, change, 1960-2017 ($UrbPop_{2017}$ / $UrbPop_{1960}$)	Urban pop, change, 2017-2050 ($UrbPop_{2050}$ / $UrbPop_{2017}$)
North America	4.8	35.9	1.77	1.19	82.8	2.11	1.30
Mesoamerica	2.4	17.5	3.44	1.29	74	5.44	1.43
Caribbean	0.6	4.3	2.11	1.10	71.2	3.78	1.23
South America	5.7	42.2	2.86	1.19	83	4.64	1.27
Americas	**13.5**	**100**	**2.38**	**1.20**	**80.8**	**3.25**	**1.30**

Source: Based on Worldometers (2017). Accessed May 2, 2017, and September 3, 2017 at http://www.worldometers.info/world-population/population-by-region/

The USA, Brazil, and Mexico are by far the most populous countries of the region. Population densities vary widely throughout the region, as do population growth rates.

Population growth rates throughout the region have generally fallen substantially since 1960. This is less true for the Caribbean subregion as a whole. Several countries' annual population growth rates have been more volatile than their subregion's overall trend: Greenland in North America, Belize in Mesoamerica, Grenada and Antigua & Barbuda in the Caribbean, and Guyana and Suriname in South America (World Bank, 2017[24]).

[24] Data available at http://data.worldbank.org/indicator/

Population trends have an important role in explaining changes in natural resources and biodiversity (Table 4.9 and Table 4.10). Population growth has been identified as a key driver of global greenhouse gasses emissions (IPCC, 2014a). However, the analysis of population growth, as an indirect driver of changes in nature and NCP needs to be completed by including the consumption patterns and life-styles considerations (Pichs, 2008, 2012).

The global middle class is expected to grow from 1.8 billion in 2009 to 4.9 billion by 2030. Much of this will occur in developing countries (including Latin America and the Caribbean) where 70% of global economic activity will emerge by 2050. With this trend comes increasing demand for energy, infrastructure, and consumer goods (Runde and Magpile, 2014; Myers & Kent, 2003).

Table 4.10. Combining population growth with per capita consumption of natural resources and assessing the level of pressures on biodiversity and ecosystems.

Table 4 10 Combining population growth with per capita consumption of natural resources and assessing the level of pressures on biodiversity and ecosystem services.

Source: Elaborated by the authors based on ECLAC (2014), CEPAL (2017), IMF (2017), UNDP (2016), The World Bank Database (2017), UNCTAD (2016), Worldometers (2017), WWF (2014, 2016); GFN (2017).

Low Population Growth / High per Capita Consumption of Natural Resources High pressures on BD resources mainly due to high per capita ecological footprint. This is a typical pattern of several industrialized countries. Critical role of international trade.	**High Population Growth / High per Capita Consumption of Natural Resources** Very high pressures on BD, due to the combined effect of increasing population / density and growing per capita ecological footprint. Critical role of international trade, and adverse implications in terms of high GHG emissions, land use changes (deforestation) and general overexploitation of natural resources.
High Population Growth / High per Capita Consumption of Natural Resources Low pressures on BD due to low population and population density, as well as low per capita ecological footprint.	**High Population Growth / High per Capita Consumption of Natural Resources** Low pressures on BD mainly due to survival reasons of growing population. Typical pattern of least developed countries and poor communities.

Population growth projections for the Americas range from around 10% (in the Caribbean) to near 30% (in Mesoamerica) between the years 2017 and 2050. At the same time, GDP projections range from 3.1 to 3.7 times in the developing regions of the Americas (around 70% in North America) in relation to the 2017 levels by 2050. Consequently, core baseline scenarios regarding the consumption of natural resources and energy in the Americas would be mainly driven by GDP growth, and population growth, as relevant drivers (Ruijven et al., 2016).

4.3.6 Human development

Analysis of the various dimensions of human development is critical for assessing the wide range of indirect drivers for changes in nature and NCP. Several social indicators and aggregated indexes may be useful for achieving that assessment purpose, including the Human Development Index (HDI) that can provide information on the share of population in extreme poverty, income distribution (e.g. Gini coefficient), educational attainment (e.g. access, literacy level), health (e.g. access to public health, health care infrastructure, expectancy of life), social expenditure / GDP (e.g. education, health), and food security (e.g. number and % of hungry people) (see Chapter 2, section 2.6).

Social inequity is still a concern for the various subregions of the Americas, with adverse implications for nature, NCP and good quality of life. On the one hand, poor people in the Americas often increase the demand pressures on nature merely to survive. On the other hand, high per capita consumption by affluent segments of the population also increases pressure on natural resources in. This discussion is very relevant in the context of the global debate on multidimensional progress (PNUD, 2016) and the SDG, particularly for key areas of social development like poverty and hunger eradication, as well as access to education, health, safe water and sustainable energy.

In 2015, Mesoamerica showed the lowest regional HDI in the Western Hemisphere, which was below the average levels for Latin America and the Caribbean countries (0.7310), the Americas (0.7418), and the world (0.7170). Haiti had the lowest country-specific HDI in the Americas (0.4930), even below the corresponding level for Sub-Saharan Africa (0.5230). Inequality Adjusted HDI was considerably lower than HDI in the Americas (by 21%), in Latin America and the Caribbean countries (by 22%) and in North America (by 11.1%) (¡Error! No se encuentra el origen de la referencia.).

Country-specific HDI values and trends indicate that most countries of the Americas rank as "very high" or "high" human development within the world community. However, four Mesoamerican and three South American countries have HDI values that rate their human development as "medium" within the world community, while Haiti's HDI falls very low in the world rankings (UNDP, 2016).

Average HDI values for all regions of the Americas improved from 2010 to 2015, representing widespread regional gains in incomes, education, and socioeconomic factors that increase life expectancy. Despite those overall improvements, HDI scores for 18 countries in the region dropped in the worldwide rankings between 2010 and 2015, indicating a failure to match gains in human development at a more international level. Of these 18 countries, half are in the Caribbean subregion.

Cuba (with 48 points) and Barbados (20 points) lead the list of countries of the Western Hemisphere where the "gross national income ranks minus HDI rank" shows positive results, indicating that their human development achievements go far beyond those derived from their gross national income. These results may be associated, for instance, with more efficient allocation of economic resources to social goals like education and health (UNDP, 2016).

Income inequality is high in the Americas overall. Most countries in the region have a degree of income inequality (reflected in low international ranks in terms of equality and high Gini coefficients) that ranks among the world's 50 most unequal nations. This is particularly true of countries in the Mesoamerican and South American subregions (Index Mundi, 2017[25]). The ratio of inequality-adjusted -HDI/HDI shows that inequality is constraining the region's societies from realizing their human development potential (¡Error! No se encuentra el origen de la referencia.).

Table 4.11. HDI and inequality adjusted HDI in the Americas (*), 2015.

Region	No. of countries	HDI 2015	No. of countries	Inequality Adjusted HDI, 2015 (IA-HDI)	IA-HDI / HDI (change in %)
North America	2	0.9200	2	0.8175	-11.1
South America	12	0.7438	12	0.5854	-21.3
Caribe	13	0.7365	5 (**)	0.5502	-20.5
Mesoamerica	8	0.7028	8	0.5345	-23.9
Americas	35	0.7418	27	0.5810	-21.0
Latin America and the Caribbean	33	0.7310	25	0.5621	-22.0
World	188	0.7170	151	0.5570	-22.3

Note:
(*) The HDI is a statistic constructed by combining a range of indicators thought to capture human potential and development: per capita income, education, and life expectancy. The inequality-adjusted HDI statistically adjusts the HDI to account for income inequality, in order to reflect the potential for human development in the absence of inequality. Higher HDI and inequality-adjusted -HDI scores indicate better conditions in these areas combined; that is, greater human well-being and potential for human well-being, respectively.
(**) Trinidad and Tobago, Jamaica, Saint Lucia, Dominican Republic and Haiti. Sources: Based on UNDP (2016).

[25] Available at https://www.indexmundi.com/facts/indicators/SI.POV.GINI/rankings/central-america

The prevalence of extreme poverty in the Americas has decreased considerably since 1981. The World Bank data show that the portion of the population of Latin America and the Caribbean living below the international "income poverty" line of $1.90 per day fell from 23.9% in 1981 to 5.6% in 2012, and that living below the international "working poor" poverty line of $3.10 per day fell from 38.0% to 12.0% over the same period (World Bank, 2017[26]).

Nevertheless, poverty in the Latin America and the Caribbean region remains a concern. First, the proportion of the population facing extreme poverty varies considerably throughout Latin America and the Caribbean at the country level. More than a quarter of the populations of El Salvador and Honduras live on less than $3.10 per day. Second, extreme income poverty in the Latin America and the Caribbean region, even at reduced levels, affects millions of people, including many children (World Bank, 2017[27]). Third, 38% of the Latin America and the Caribbean region's population is socioeconomically vulnerable due to a persistent inability to enter the middle class (PNUD, 2016). Fourth, the recent worldwide economic slowdown exacerbates this susceptibility.

The percentage populations living in poverty in 2012 was approximately 26.9% in Latin America, 40.6% in Mesoamerica, and 21% in South America (CEPAL, 2014). Around 72 million people exited the condition of income-poverty during 2003-2013 in Latin America; however, 25-30 million people are at risk of falling into that condition again as a result of economic vulnerability and social fragility (PNUD, 2016).

Poverty not only affects the developing countries in the Americas. The percentage of poor people recently reached 13.9% in the USA population (43.1 million people)[28]; and those living in households below statistics Canada's low income threshold represented 9.7% in 2013; incidence of low income tended to be higher among children, seniors, and persons in single-parent families (Lammam & MacIntyre, 2016).

Historically, the needs and priorities of indigenous peoples in the Americas have been largely ignored, mainly affecting indigenous women. This situation has started to change in recent past. By 2010, about 45 million indigenous people (8.3% of the regional population) lived in Latin America, compared with an estimated 30 million in 2000, an increase that is partially a result of population growth but also from the greater visibility of this population in the national censuses. On average, without distinguishing educational levels, the labor income of non-indigenous and Afro-descendant men quadrupled those of indigenous women and almost doubled those of Afro-descendant women. Between 2009 and 2013, around 235 conflicts were identified in Latin America, which were generated by projects of extractive industries (mining and hydrocarbons) in indigenous territories (CEPAL, 2016).

The population of American indians and Alaska natives in the USA, including those of more than one race, comprised approximately 2.0% of the total population (6.6 millions) in 2015[29]. Data from the National Household Survey in Canada show that 1,400,685 people had an Aboriginal identity in 2011, representing 4.3% of the total population[30].

Another set of broader societal factors deserving special consideration when dealing with the implications of social development on biodiversity and ecosystem services include worldviews and culture (attitudes to environment/sustainability/equity), life-styles (including diets), and societal tensions and conflict levels.

Culture in the form of the values, norms, and beliefs of a group of people can act as an indirect driver of ecosystem change by affecting environmentally relevant attitudes and behaviours (IPBES, 2016).

Biodiversity and linguistic diversity are threatened globally. They are declining at different rates in different regions, with the most rapid losses in linguistic diversity occurring in the Americas, which is in parallel to biodiversity loss (Maffi, 2005; Harmon & Loh, 2010; Gorenflo et al., 2012).

---

[26] Available at: Povcal Net, Online Database – http://go.worldbank.org
[27] Available at http://povertydata.worldbank.org/poverty/region/LAC
[28] According to data from the Center for American Progress (2017). Available at https://www.census.gov/content/dam/Census/library/publications/2016/demo/p60-256.pdf., quoted by https://talkpovertry.org/basics/
[29] Vintage 2015 Population Estimates: http://nativenewsonline.net/currents/u-s-census-bureau-native-american-statistics/
[30] http://www12.statcan.gc.ca/nhs-enm/2011/as-sa/99-011-x/99-011-x2011001-eng.cfm

In this context, indigenous and local communities' traditional knowledge provides a comprehensive reflection of prevailing conditions and other key inputs and incorporates methods and approaches that capture holistic values that people place on nature, while internalizing principles and ethical values specific to their world views and realities (Illescas and Riqch'arina, 2007; Medina, 2014, quoted by IPBES, 2016).

Traditional ecological knowledge can be found all over the world, particularly within indigenous traditions across diverse geographical regions from the Artic to the Amazon, and represents various understandings of ecological relationships, spirituality, and traditional systems of resource management (Alexander et al., 2011). In recent decades, resource managers have gradually begun to embrace the usefulness of applying that knowledge to contemporary stewardship issues in various parts of the world.

Indigenous peoples in multiple geographical contexts, including the Americas, have been pushed into marginalized territories that are more sensitive to environmental challenges, in turn limiting their access to food, cultural resources, traditional livelihoods and place-based knowledge. All this disrupts their ability to respond to environmental changes and undermines aspects of their socio–cultural resilience (Ford et al., 2016) (Box 4.7).

The broad ways in which indigenous knowledge and experiences are framed mirror common portrayals of indigenous peoples as "victim–heroes"; "victims" through the framing that indigenous peoples are highly vulnerable and "heroes" through the framing that indigenous peoples possess knowledge that can help address the problem (Ford et al., 2016). The complexity and diversity of indigenous experiences and their understanding and responses to environmental challenges are not well captured in many of the cases where indigenous content is documented by peer review literature.

Some studies identify the ongoing effects of colonialism, marginalization, power relations, land dispossession and land rights to be central to understanding the human dimensions of global environmental change for indigenous peoples in diverse contexts (Ford et al., 2016).

Box 4.7. Indigenous and local knowledge and values: Implications for natural resources management

The Americas are populated by many indigenous nations, from the Artic to Patagonia, with a variety of cultures and languages that have developed many different socio-economic systems (nationally and locally). Increasing numbers of historically marginalized groups are joining transnational networks and alliances that promote indigenous mobilization and demand recognition and rights from their respective nation-states and the international community. These rights include protection of and control over their property and possessions (like territories, resources, material culture, genetic material, and sacred sites), practices (cultural performances, arts, and literature), and knowledge (cultural, linguistic, environmental, medical, and agricultural). By linking issues of representation, recognition, resources, and rights, these movements engage and often challenge current theories of culture, power, and difference in sociocultural anthropology (Hodgson, 2002). Indigenous and local knowledge are expressions of social capital and may act as a driver of biodiversity and ecosystem services supply because of direct influences on land use change (direct influences), as well as its ability to modify the influence of other drivers (interactive influences). Some cases illustrating the role of ILK as drivers of land use change in the Americas, hence on biodiversity and ecosystem services, are presented below:

1. The Isobore Sécure National park and indigenous territory case in Bolivia *(*McNeish, 2013). In August 2011, 2000 marchers left the city of Trinidad, the lowland regional capital of the department of Beni, to follow a route that would take them 66 days and 600 kilometers of walking to the capital city of La Paz. The central demand of the protest march was founded on the cessation of a road-building project planned to go through the Isobore Sécure National Park and Indigenous Territory. Following a series of meetings between the protesters and the president, the government agreed to pass a legal decree on 24 October 2011 guaranteeing that the road would not pass through the Isobore Sécure National Park and Indigenous Territory. Furthermore, the law stated that the Isobore Sécure National Park and Indigenous Territory would be protected by the state as an 'intangible' territory, effectively making the territory out of bounds for all forms of future state or development projects.

2. Shrimp farming versus mangroves in coastal Ecuador (Veuthey & Gerber, 2012). Over the last two decades, the global production of farm-raised shrimps has increased at a faster rate than any other aquacultural product, leading to massive socio-ecological damages in the mangrove areas where shrimp farming often takes place. Consequently, an increasing number of conflicts pitting coastal populations against shrimp farmers have been reported; although, very few conflicts have been studied in detail. According to the authors, the development of shrimp farming can be understood as a modern case of enclosure movement whereby customary community mangroves are privatized for the building of shrimp ponds. As a result, local mangrove-dependent populations – especially women – mobilized and protested against a form of ecologically unequal exchange. While only some mangroves could be saved or reforested as a result of the movement, women's mobilization has had the unexpected effect of challenging gender relations in their communities.

3. Oil frontiers and indigenous resistance in the Peruvian Amazon (Orta-Martínez & Finer, 2010). The Peruvian Amazon is culturally and biologically one of the most diverse regions on Earth. Since the 1920s oil exploration and extraction in the region have threatened both biodiversity and indigenous peoples, particularly those living in voluntary isolation. Modern patterns of production and consumption and high oil prices are forcing a new oil exploratory boom in the Peruvian Amazon. While conflicts spread on indigenous territories, new forms of resistance appear and indigenous political organizations are born and become more powerful.

4. Indigenous land and deforestation control in Amazon (Nepstad et al., 2006). Indigenous lands occupy one-fifth of the Brazilian Amazon. Analyses of satellite-based maps of land cover and fire occurrence in the Brazilian Amazon compared the performance of large (>10,000 ha) un-inhabited (parks) and inhabited (indigenous lands, extractive reserves, and national forests) reserves. Reserves significantly reduced both deforestation and fire. There was no significant difference in the inhibition of deforestation or fire between parks and indigenous lands, but uninhabited reserves tended to be located away from areas of high deforestation and burning rates. In contrast, indigenous lands were often created in response to frontier expansion, and many prevented complete deforestation despite high rates of deforestation along their boundaries.

4.4 Direct anthropogenic drivers

4.4.1 Habitat degradation and restoration

Nature of the driver, its recent status and trends, and what influences its intensity

Habitat degradation includes land conversion and intensification of croplands and rangelands; wetland drainage and conversion; construction of roads, dams, pipelines, and transmission lines; sprawl; pollution, and resource extraction. Physical alterations of freshwater habitats also include change in hydrological regime (flow regime and water withdrawals). Marine environment degradation is increasing in some areas with increased shipping and bottom trawling, coastal construction (ports, marinas, housing and other development, and pollution with various forms of sediment and chemical discharges. Aquaculture (farming of marine flora and fauna) also can contribute to habitat degradation (for ponds, access and infrastructure; for feed: fishing to produce fish meal, hormone and antibiotic additives; discharges in the form of fecal pollution, etc.). Pollution as a driver of change will be discussed in the section 4.4.2.

Habitat loss and degradation are considered the greatest threats to biodiversity (Wilcove et al., 1998, Sala et al., 2000, Hanski et al., 2013, Murphy & Romanuk, 2014; Haddad et al., 2015; Newbold et al., 2015). Worldwide, nearly half of tropical dry forests, temperate broadleaf forests, and temperate grasslands, savannas, and shrublands have been converted to human uses (Hoekstra et al., 2005). Land use change affects biodiversity and ecosystems not only by reducing population sizes and movements, but also by reducing habitat area, increasing habitat isolation, and increasing habitat edge (Haddad et al., 2015). Reducing area or increasing isolation decreases both species persistence and species richness (Haddad et al., 2015).

Forests covered 1.6 billion hectares of land in the Americas, which is approximately 41% of its land area and 40% of worldwide forest area (Yearbook FAO Statistical, 2013). This forest includes 722 million hectares of relatively undisturbed old-growth forest, 57 million hectares of planted forest, and 818 million hectares of forest that regenerated after human disturbance. From 1990 to 2015, forest area expanded in North America by nearly three million hectares and the Caribbean by more than two million hectares but declined in Central America by nearly seven million hectares and in South America by more than 88 million hectares (Keenan et al., 2015). Approximately 34% of forest area is protected in South America (where the percentage of protected forest area doubled from 1990-2005) and less than 9% of forest area is protected in North America (in accordance with the IUCN definition, excluding categories V and VI) (Morales-Hidalgo et al., 2015). Brazil has a much higher proportion of its forest protected (41.8%, 206 million hectares) than any other country and the USA has protected the second greatest forest area (33 million hectares, 10.6% of forests; Morales-Hidalgo et al., 2015).

Conversion to croplands and pasturelands is the main driver of terrestrial habitat change in the region. In 2013, agriculture covered 1.23 billion hectares of land in the Americas, which is approximately 32% of its land area and 25% of the worldwide agricultural land (Yearbook FAO Statistical, 2013). This agriculture included 828 million hectares of permanent meadows or pastures and rangelands used for livestock grazing (68%), 28 million hectares of permanent crops, and 370 million hectares of arable land (~2%), which includes land covered by temporary crops, pasture, or hay meadows (~30%). Conversion patterns differ among subregions. Most land conversion in Mesoamerica and North America occurred more than one century ago, whereas in South America most occurred within the last century. Since 1961, the area of agricultural land has increased by 13% across the Americas, which is the net result of a 40% increase in South America, a 29% increase in the Caribbean, an 11% increase in Central America, and a 9% decrease in North America. From 2001 to 2013, 17% of new cropland and 57% of new pastureland replaced forests throughout Latin America (Aide et al., 2013). Cropland expansion from 2001 to 2013 was less (44.27 million hectares) than pastureland (96.9 million hectares), but 44% of cropland in 2013 was new, versus 27% of pastureland, revealing row crop expansion. Most cropland expansion was into pastureland within agricultural regions of Argentina, Brazil, Bolivia, Paraguay, and Uruguay (Graesser et al., 2015, Volante et al., 2015). Commodity crop expansion, for both global and domestic urban markets, follows multiple land change pathways entailing direct and indirect deforestation, and has various social and environmental impacts (Meyfroidt et al., 2014, see Chapter 2, section 2.2.1).

Agricultural practices associated with land conversion significantly change biogeochemical cycles contributing to pollution of terrestrial and aquatic ecosystems and to climate change (sections 4.4.2 and 4.4.3). Each year, land conversion results in emissions of approximately one billion metric tonnes of carbon (1 Pg C per year), which is 10% of emissions from all human activities (Friedlingstein et al., 2010). Soil carbon losses also diminish crop yields and degrade water quality. Nitrogen fertilization also contributes to climate change by emitting the greenhouse gas of nitrous oxide (Compton et al., 2011; Sutton et al., 2011; Keeler et al., 2016). In the Americas, approximately 23 million tonnes of nitrogen fertilizer and 22 million tonnes of phosphorus (phosphate + potash) were consumed in 2013; and about 52 million hectares of land were under irrigation. Increasing anthropogenic nitrogen inputs are also likely driving loss of diversity (Bobbink et al., 2010) and polluting freshwater supplies (section 4.4.2). Nutrient imbalances due to agriculture are related to depletion or accumulation depending on the balance between inputs and outputs of nutrients. Nitrogen depletion occurred in the southern parts of South America (e.g. Argentina), the Amazon region, Central America, and some parts of the Midwest of the USA, partially attributable to the high crop yields (Liu et al., 2010). Soil nitrogen depletion occurs regardless of how high the nitrogen input once crop nitrogen uptake, along with other nitrogen losses, exceeds the inputs (Liu et al., 2010).

Croplands also affect migratory species through habitat degradation and pesticide use along their migratory routes (e.g. neotropical migratory birds like dicksisels, bobolinks, and Swainson’s hawks) (Basili & Temple, 1999; Hooper et al., 2002; Lopez-Lanus et al., 2007). Habitat conversion leads to not only many native species losses, but also to gains in some exotic species (section 4.4.4). Exotic species are often introduced for particular human uses and are not necessarily functionally equivalent to the native species they displace

(Wardle et al., 2011).

Urbanization can also directly and indirectly threaten biodiversity and services from surrounding ecosystems. In 2016, while the degree of urbanization worldwide was around 54%, it was around 80% in the Americas. In Latin America and the Caribbean, the urbanization rate has declined over the past six decades (UN, 2014). Cities in Latin America exhibit extreme social and economic differences, which generate a complex mosaic of urban settlement structures and ecosystem management systems. In addition, conservation of ecosystems and biodiversity, and ecosystem services provisioning, are not prioritized in urban planning (Pauchard & Barbosa, 2013). Direct impacts include land occupation by buildings and roads. Indirect impacts result from the provisioning of services to urban populations, like food, building materials, energy, water, and other resources. This requires infrastructure such as dams, pipelines, transmission lines, and roads, timber harvesting, and land cover conversion for grazing and cropping. (e.g. McDonald et al., 2014; Bhattacharya et al., 2012). Roads help deliver benefits from where they are supplied to where they are demanded and consumed. However, they also threaten biodiversity (Laurance et al., 2014) by fragmenting habitat and facilitating resource extraction activities like cropping; grazing; timber harvesting and extraction of water, minerals, oil, and gas. For example, over the last 60 years, there have been at least 238 notable oil spills along mangrove shorelines worldwide. In total, at least 5.5 million tonnes of oil has been released into mangrove-lined, coastal waters, oiling possibly up to around 1.94 million hectares of mangrove habitat and killing at least 126,000 hectares of mangrove vegetation since 1958 (Duke, 2016). Mangroves and other coastal “blue carbon” ecosystems also have high ecosystem carbon stocks and are undergoing significant conversion at a great cost in terms of greenhouse gas emissions, as well losses of other important ecosystem services (Kauffman et al., 2016).

Despite declines in the density of species, cities can have unique assemblages of plants and animals and retain some endemic native species, thus providing opportunities for regional and global biodiversity conservation, restoration and education (Aronson et al., 2014). Habitat conversion has also resulted in increases in food, mineral, timber, and energy production. For example, global cereal production has more than doubled since 1960 (Tilman et al., 2002; Wik et al., 2008). Few studies have weighed such benefits against the costs of habitat degradation described above. In some cases, however, the financial costs of habitat conversion for non-provisioning ecosystem services, like carbon storage and sequestration, can outweigh the benefits of conversion for supply of provisioning services (Nelson et al., 2009)

The intensity of land degradation depends on indirect drivers (section 4.3), like governance (zoning, incentive policies, management policies), social development (education, technology), economic development (markets, trade, technology, land tenure, corporate pressures), and interactions among land degradation and other direct drivers, including climate change and changes in fire regimes. With economic development, human diets have shifted toward more meat and dairy consumption (Foley et al., 2011, Tilman et al., 2011). Continuing this trend in coming decades would require further pasture expansion, intensification of livestock production, or both. Maintaining or increasing future food, energy and water production without compromising biodiversity and ecosystem services can involve multiple strategies, including land sharing and land sparing (Fisher et al., 2014); closing yield gaps on underperforming lands (Mueller et al., 2012); improving efficiency of agricultural input application, reducing food waste (Foley et al., 2011) and changing diets (Tilman et al., 2011, Tilman & Clark, 2014; Vranken et al., 2014).

After abandonment from human uses, some habitats gradually recover while others fail to do so (Benayas et al., 2009; Jones & Schmitz, 2009; Barral et al., 2015). Over the past 15 years, total global pasture area decreased by 2%, with much of that land likely abandoned, rather than converted to other agriculture (Poore, 2016). There is substantial potential for biomass recovery of Neotropical secondary forests, with most forests recovering 90% of biomass in less than a century (Poorter et al., 2016). Based on well documented evidence of the negative impacts of deforestation on surface water quality (Baker et al., 2004; Scanlon et al., 2007) it is possible that the reverse of deforestation will improve stream water quality in freshwater systems, especially with active forest restoration.

Even with active ecosystem restoration, however, it is rarely possible to fully restore lost biodiversity and ecosystem services (Benayas et al., 2009). Habitat restoration often significantly increases biodiversity and ecosystem services above levels observed in degraded ecosystems, but levels of biodiversity and ecosystem services in restored ecosystems often remain significantly lower than levels in reference remnant ecosystems. Compared with reference ecosytems, recovering ecosystems exhibit annual deficits of 46–51% for organism abundance, 27–33% for species diversity, 32–42% for carbon cycling and 31–41% for nitrogen cycling (Moreno-Mateos et al., 2017). Although degradation of ecosystems is ongoing, there is also a significant increase in conservation and restoration efforts in the Americas (Wortley et al., 2013; Echeverría et a. 2015). Some examples of restoration of terrestrial and freshwater ecosystems are presented in Box 4.8 and Box 4.9.

Box 4.8. Examples of restoration initiatives in the Americas – Great Lakes

The five Laurentian Great Lakes – Superior, Huron, Michigan, Erie and Ontario – comprise 20% of the world's available freshwater supply. The Great Lakes cover an area of about 246 million km². The draining basin extends from roughly 41 to 51° N, and from 75 to 93° W, and includes parts of eight USA states and two Canadian provinces. Human activity has had deleterious impacts on the Great Lakes ecosystem. The logging boom of the late 1800s altered the basin's hydrologic regime. Shipping traffic introduced non-native species and untreated waste discharge of nutrients and other chemical pollutants led to a virtual ecological collapse in the mid-1900s (Rankin, 2002).

Since 2009, the Great Lakes have been the focus of a major restoration initiative by the USA government (expenditures of greater than $1 billion over five yers), targeting invasive species, nonpoint run-off, chemical pollution, and habitat alteration. The current initiative specifically targets key classes of environmental stressors that were identified through a planning process involving numerous government agencies and environmental groups (Allan et al., 2013). For example, Great Lakes Restoration Initiative resources have been used to double the acreage enrolled in agricultural conservation programs in watersheds where phosphorus runoff contributes to harmful algal blooms in western Lake Erie, Saginaw Bay and Green Bay (https://www.glri.us).

The Great Lakes sand dunes constitute the most extensive freshwater dunes in the world, covering over 1,000 km² in Michigan alone (Albert, 2000). In the region, traditional dune restoration efforts involving monoculture plantings of *A. breviligulata* (American beach grass) restore many measures of diversity and ecosystem function over the past 20-30 years (Emery & Rudgers, 2009). Plant and insect diversity, vegetation structure (plant biomass and cover), and ecological processes (soil nutrients and mycorrhizal fungi abundance) in restored sites were similar to reference sites. Differences were mostly attributed to the relative age of the sites, where the younger sites supported slightly lower plant diversity and mycorrhizal spore abundance than older sites (Emery & Rudgers, 2009).

Box 4.9 Examples of restoration initiatives in the Americas – Tropical forests and pastures

The presence of degraded areas, many of them already abandoned, in almost all types of land use, generate further degradation and impacts on natural remnants, like effects on pollinators through uncontrolled application of pesticides. The persistence of these practices will lead to the emergence of additional degraded areas. Two different and coordinated actions could be considered in order to provide potential solutions for these environmental problems: 1) actions to avoid, stop, minimize or reverse the ongoing environmental degradation (e.g. fire management, erosion control, reduction of pesticide use, among others) which could be generically called sustainable management practice, and 2) specific actions for the recovery of already degraded areas, that is, restoration. Productive and environmental landscape optimization, in addition to the actions forementioned, is also intended to change land-use economic practices, locally increasing productivity, thereby reducing pressures to use areas that have more value for conservation. Effective actions have been taken in many regions of the world that correspond to sustainable management practices (FAO et al., 2011; FAO, 2011 and 2013), rehabilitation (Buckingham & Hanson, 2015) and restoration of degraded areas (Nellemann & Corcoran, 2010; Goosem & Tucker, 2013; Hanson et al., 2015).

In the Americas there are already important examples of the successful implementation of sustainable management practices (e.g. ITTO, 2011; Calle et al., 2012; Calle & Murgueitio, 2015; FAO, 2013), rehabilitation (e.g. Brancalion et al., 2012), and restoration (e.g. Calvo-Alvarado et al., 2009; Rodrigues et al., 2009, 2011; Murcia & Guariguata, 2014; Hanson et al., 2015). Restoring distinct vegetation types that have very different levels of resilience, species richness and complexity of interactions and are inside landscapes with different degrees of fragmentation have demanded different methods. Although the degree of success achieved for each one varies between vegetation types and socioeconomic conditions considered, there are already examples in Brazil where restoration in large-scale and high-biodiversity tropical forests have been achieved (Rodrigues et al., 2011) and whose principles could be adapted to other vegetation types and countries. An example is the intensive silvopastoral systems, which have been implemented in Colombia (Calle et al., 2012). Livestock grazing, a common practice in the Americas and around the world, results in soil compaction, soil erosion, reduction of water infiltration, and silting of springs and streams. This degraded land condition can maintain very few animals and produces less income. Grazing also favors continuous land abandonment and migration, inducing deforestation to create new pastures. Converting extensive pastures to intensive silvopastoral systems allowed for, in 4-5 years, increases in production, productivity, and rural incomes and jobs, as well as the elimination of all sources of degradation. This change resulted in increases of environmental services and rural biodiversity and allowed for the release of farm margins to be used for forest restoration or rehabilitation.

North America

Oil and gas development in Alaska and Canada has focused on tundra in North America since the 1960s (Maki et al., 1992). Its effects on birds and mammals can extend beyond the area occupied by oil and gas industrial infrastructure. Cameron et al. (2005) found that calving caribou abundance was lower within 4 km of roads in an oil and gas development area and declined exponentially with road density. With increasing infrastructure, high-density calving shifted inland, despite the lower forage biomass there (see also Wolfe et al., 2000). Similarly, passerine bird nests are at greater predation risk within 5 km of infrastructure (Liebezeit et al., 2009; see also Weiser & Powell, 2010). Substantial tundra habitat changes are expected with climate change that may have substantially greater impacts on habitat than human infrastructure, including increases in shrub-dominated ecosystems and changes in wetland abundance and distribution (section 4.4.3).

Boreal forest disturbance (tree cover loss), due largely to fire and forestry, was globally the second largest in both absolute and proportional extent from 2000-2010 (Hansen et al., 2013). North America presented the higher overall rate of forest loss in comparison with other boreal coniferous and mountain ecozones in the world. In boreal forest, fire is the primary natural disturbance (see also section 4.5). Fire creates a complex mosaic of stands of varying age, composition, and structure, within which other disturbances and processes interact. Thus, it has been suggested to attenuate the impacts of logging on a managed landscape; logging should create patterns and processes resembling those of fire. However, logging has already shifted forest age-class distributions to younger stands, with a concurrent decrease in old-growth stands, and is quickly forcing the landscape outside of its long-term natural range of variability (Cyr et al., 2009). Fire severity is a key component of regeneration trajectory (Johnstone et al., 2010). Increases in boreal fires severity with climate warming may catalyze shifts toward deciduous-dominated forests, altering landscape dynamics and ecosystem services (see also sections 4.4.3 and 4.5). Besides climate impacts, other anthropogenic environmental changes like changes in biogeochemical cycles (section 4.4.2) and exotic invasive species (section 4.4.4) can interact with heat and drought (Millar & Stephenson, 2015) to negatively affect temperate and boreal forests.

The traditional fire knowledge of many native American cultures of North America was lost during European settlement. Many groups experienced declining the traditional fire knowledge systems abruptly and for several generations as most indigenous peoples in the subregion were forced from their ancestral lands, punished for speaking their native languages, and forbidden to use fire in open native vegetation. Some tribes, however, retained enough traditional fire knowledge although they did not practice traditional burning continuously on the landscape (Huffman, 2013).

Many temperate forests have at some time been used for agriculture. Large-scale deforestation first occurred during the 18th-19th centuries (Flinn & Vellend, 2005). Particularly across northeastern North America, phases of forest clearance were followed by agricultural use, agricultural abandonment, old-field succession, and then forest regeneration. Generally, species richness within forest stands (alpha diversity) remains lower in recent compared to ancient forests, even when recent forests are decades or centuries old (Flinn & Vellend, 2005). This biotic homogenization is legacy of human land-use that may endure for decades if not centuries (Leps & Rejmánek, 1991; Vellend, 2007; Thompson et al., 2013; Deines et al., 2016). Additionally, fire once shaped many North American ecosystems, but Euro–American settlement and 20th-century fire suppression drastically altered historic fire regimes, shifting forest composition and structure (McEwan et al., 2011; Ryan et al., 2013).

Earlier in the 20th century, USA land cover was on a trajectory of forest expansion after agricultural abandonment (Drummond & Loveland, 2010). The expansion of forest cover since 2000 has been offset by forest loss, with forest loss evenly divided among cropland, pasture and urban/suburban land (Masek et al., 2011). The potential for forest regeneration has slowed, however, because forest conversion to urban/suburban land is less reversible. In addition, in some regions, like the eastern USA, tree cover has declined because forest harvest rates have outpaced reforestation (Drummond & Loveland, 2010, Masek et al., 2011, Hansen et al., 2013). Currently, according to Hansen et al. (2013) the northwestern USA is an area of intensive forestry, as is all of temperate Canada. Land-use pressures significantly impact the extent and condition of eastern USA forests, causing a regional-scale decline in tree cover, mainly from urban expansion. Annual forest loss accelerated from approximately 56,000 hectares from 1973-1980 to 90,000 hectares by 1992-2000 (Drummond & Loveland, 2010).

Prairie grasslands dominated central North America for millennia, until the mid- to late-1800s when European settlers converted them to croplands and rangelands (Ellis et al., 2010). North American grasslands are now some of the planet's most heavily converted ecosystems (Isbell et al., 2015). As a result of this dramatic habitat loss and fragmentation, these grasslands are rapidly losing plant species (Leach & Givnish, 1996; Wilsey et al., 2005). Even more notable, nearly all of them have lost their keystone herbivores, including bison and elk. For example, during the mid-1800s, bison populations declined from tens of millions to a few thousand individuals (Knapp et al., 1999). Since that time, bison numbers have increased to more than 100,000 individuals in public and private herds that are maintained for prairie restoration or meat production. Rangeland degradation in the west, grassland conversion to croplands, and afforestation of old fields in the east have together caused North American songbirds to sharply decline in recent decades (Brennan & Kuvlesky, 2005). Increased use of prescribed fire and grazing as sources of disturbance, and sowing of seeds to overcome dispersal limitation in fragmented agricultural landscapes, have improved prairie grassland restoration, preventing woody encroachment and restoring native plant diversity (Martin et al., 2005).

A second wave of conversion of remaining fragments of North American grasslands to croplands, including 530,000 hectares from 2006-2011 in the upper Midwestern USA alone, has resulted from the recent doubling of crop prices following increased demand for biofuel feedstocks. These grasslands escaped conversion until only recently because they are particularly vulnerable to erosion and drought, or because they are adjacent to wetlands (Wright & Wimberly, 2013). The relationship between biofuel production and food prices is controversial in the scientific literature and depends on several factors as increased demand, decreased supply, and increased production costs driven by higher energy and fertilizer costs. Disentangling these factors and providing a precise quantification of their contributions is difficult but there is a convergence that analysis should include short and long-run effects, type of crops and technology (first or second-generation biofuels) as different biofuels have different impacts (Rathman et al., 2010; Ajanovic, 2011; Mueller et al., 2011; Zilberman et al., 2013; Koizumi, 2015; Filip et al., 2017).

Drylands in North America (the hot Sonoran, Mojave, and Chihuahuan deserts and the cool Columbia Plateau, Great Basin, and Colorado Plateau deserts) have experienced moderately low to high appropriation of land by humans; degraded to very degraded fire cycles; very high to extremely high habitat fragmentation; and habitat losses between 2000 and 2009 of up to 11% (Hoesktra et al., 2010). Intensive cropping in many areas has lowered water tables and the amount of fertilized and salinized soil, leading to land abandonment with ensuing invasion by exotic annual grasses and reduced biodiversity and ecosystem function (Gelt, 1993). Most of these lands have been grazed by livestock since the early 1800s, and as most current grasses did not evolve with large mammal herds, this grazing has caused native species losses, altering plant and animal community composition, (Mack & Thompson, 1982). Climate change models are predicting higher temperatures and reduced precipitation for North American drylands (Cook et al., 2004; Christensen et al., 2007), likely leading to long-term declines in soil moisture, which will negatively affecting shallow-rooted plants (Fernandez & Reynolds, 2000; Munson et al., 2011; Wertin et al., 2015). Increasing carbon dioxide loss of grass, and altered climate and fire regimes favor woody plant encroachment, further reducing biodiversity and affecting animals that depend on native plants that are lost (Archer et al., 1995). Grasses are vital to these ecosystems; they form the base of the food web, providing forage for livestock and small mammals, promoting soil carbon sequestration, stability and fertility and thus their loss affects ecosystem function (Sala & Paruelo, 1997). These landscapes are also seeing dramatic increases in soil surface disturbance from recreation and energy and mineral exploration and extraction (Weber et al., 2016). Disturbance of the soil surface compromises the cover and function of biological soil crusts, a community of organisms that are critical to water, nutrient, and carbon cycles in drylands (Weber et al., 2016) and they may not return to their pre-disturbance state or function (Concostrina-Zubiri et al., 2014). Reduction in plant and biocrust cover increases soil erosion, which itself directly drives biodiversity loss and alters ecosystem function. Erosion reduces source soil carbon and nutrients (e.g. Neff et al., 2008; Belnap & Büdel, 2016; Weber et al., 2016; Ahlström, 2015); increases dust deposition on nearby snowpacks, which reduces the amount of water entering major rivers (Painter et al., 2010); and threatens human economic, health, and social well-being (Fields et al., 2009). Roads, pipelines, transmission lines, vegetation change, and energy developments continue to heavily fragment and degrade many drylands, especially the Mojave and Great Basin deserts (Knick et al., 2003; Hoesktra et al., 2010).

The wetlands of North America include many different wetland types, ranging from the expansive peatlands of boreal Canada and Alaska to the seasonally flooded marshes of the subtropical Florida Everglades. Wetlands of North America continue to be threatened by drainage for agriculture and urban development, extreme coastal and river management, water pollution from upstream watersheds, peat mining, waterfowl management, and more recently climate change. From 1780-1980, from 65 to 80% of wetlands in Canada were lost, while 53% of wetlands in the continental USA were lost (Mitsch & Hernandez, 2013). The middle Atlantic coastal plain experienced vast land cover change compared with other Eastern USA ecoregions, ranking third in the proportion of area changed. Two of the dominant land-cover types, forest and wetlands, experienced considerable net change (USGS, 2016). Urban development almost always increases in area, as it tends to be permanent, whereas other land-cover types, like forest, agriculture, wetlands, and mechanically disturbed lands, may fluctuate in area as part of cyclic land-use changes (USGS, 2016). Probably as a result of enforcing Clean Water Act requirements to mitigate wetland losses, as well as program such as the Wetlands Reserve Program (Wiebusch & Lant, 2017), wetland restoration and creation may have partially offset losses in rural and suburban areas since the mid-1980s (Mitsch & Hernandez, 2013).

North America contains some of the most urbanized landscapes in the world. In the USA and Canada, approximately 80% of the population is urban (Kaiser Family Foundation, 2013 in McPhearson et al., 2013). Population growth combined with economic growth has fueled this recent urban land expansion. Between 1970 and 2000, urban land area expanded annually by 3.31% (Seto et al., 2011), which was mostly cropland and forest conversion (Alig et al., 2004), creating unique challenges for conserving biodiversity and maintaining regional and local ecosystem services. Urban areas in the USA could increase by 79% by 2025, which would mean that 9.2% of USA land will be urban (Alig et al., 2004). A large portion of this increase is expected in coastal areas where populations will be exposed to issues associated with predicted sea level rise. Changes in development density will have an impact on how populations are distributed and will affect land use and land cover. Some of the projected changes in developed areas will depend on assumptions about changes in household size and how concentrated urban development will be. While higher population density means less land is converted from forests or grasslands, it can result in larger extents of paved areas and an increase in low-density exurban areas, which will lead to a greater area affected by development and increase commuting times and infrastructure costs (Brown et al., 2014).

Mesoamerica

Drivers of change in biodiversity and ecosystem function in Mesoamerican drylands (Sonoran and Chihuahuan deserts) are similar to those in North America, though they differ in relative importance (CONABIO, 2014). Livestock have grazed Mexican deserts and semi-deserts for hundreds of years. Again, lack of resistance to this herbivory has altered plant community composition, decreased native species cover, and altered nutrient, carbon, and hydrologic cycles. (Mack & Thompson, 1982). Climate models predict warmer temperatures and reduced precipitation for this region (Cook et al., 2004, Christensen et al., 2007). These changes, along with natural drought will cause loss of grasses and other shallow-rooted plants (Fernandez & Reynolds, 2000; Moreno & Huber-Sannwald, 2011) and facilitate woody plant encroachment, which is already underway (Archer et al., 1995). Loss of grasses will reduce food availability for livestock and wildlife, reduce an already limited soil carbon sequestration, reduce limited soil nutrients, alter plant and animal community composition and change ecosystem functions (Sala & Paruelo, 1997). Loss of biological soil crusts[31] and plant cover reduction with soil disturbance negatively influences water, nutrient, and carbon cycles and increases soil erosion in these ecosystems (Weber et al., 2016). Disturbed biological soil crusts may not recover to a pre-disturbance state, altering their ecosystem role (Concostrina-Zubiri et al., 2014). Grazing, cropping, energy and mineral exploration and development, and recreation are the major drivers of land degradation of Mexican deserts and semi-deserts (Sarukhan et al., 2015; Sala et al., 2000). These changes generally result in loss of biological soil crust and plant cover, resulting in soil erosion, which is a major issue in Mexican deserts and semi-desert areas (Balvanera et al., 2009). Hoesktra et al. (2010) report that these areas have experienced moderately low to moderate appropriation of land by humans, fire cycles that are degraded, very high to extremely high fragmentation, and up to 3.3% habitat losses between 2000 and 2009.

Mesoamerican forests are the third largest among the global biodiversity hotspots and are one of the most endangered ecosystems in the tropics (Sánchez-Azofeifa et al., 2014) due to high rates of forest loss and fragmentation (Chacon, 2005).

Drivers of change in Mesoamerican tropical dry forests are both negative and positive, but they still contribute to significant forest loss. Dry forests now exist as fragments of what was once a large, contiguous forest extending from Mexico to northern Argentina. The timber industry, indigenous fuel–wood

[31] Biological soil crusts result from an intimate association between soil particles and cyanobacteria, algae, microfungi, lichens, and bryophytes (in different proportions) which live within, or immediately on top of, the uppermost millimeters of soil. Soil particles are aggregated through the presence and activity of these biota, and the resultant living crust covers the surface of the ground as a coherent layer.

extraction, and cattle ranching expansion are the main drivers of dry forest loss (Fajardo et al., 2005; Calvo-Alvarado et al., 2009). These forests now cover 519,597 km^2 across North and South America. Mexico contains the largest extent at 181,461 km^2 (38% of the total), although it remains poorly represented within protected areas (Portillo-Quintero & Sánchez-Azofeifa, 2010).

In general, tropical dry forest area in Mexico is declining, with cattle ranching driving most of this deforestation, particularly along the Pacific coast (Sanchez-Azofeifa et al., 2009), even though the forest loss rate in Mexico was halved between 2010 and 2015 (Keenan et al., 2015). Unfortunately, the protected tropical dry forest in Costa Rica represents less than 1% of the total extent of this ecosystem in the Americas and is continentally less significant. Low extent and high fragmentation of dry forests in Guatemala, El Salvador, and Nicaragua mean that these forests are at high risk from human disturbance and deforestation.

There are many wetlands and freshwater systems in Mesoamerica that are each integral to a system of life, culture, a means of economic support and habitat. Tourism income represents 20.4% of the foreign earnings in Mesoamerica (Agencia EFE, 1998). The location and topographic complexity of Mesoamerica makes it unique in its water availability, with an average of 27,200 m^3 inhabitants per year. The World Meteorological Organization cites that Mesoamerican countries have few real problems with water supply, using on average less than 10% of the available water resources. However, countries like Mexico, Guatemala and El Salvador experience water shortages (IUCN 1999, https://portals.iucn.org/library/efiles/documents/1999-012.pdf). In Mexico, water shortages occur because water resources are not located close to human settlements, producing an imbalance between supply and demand and leading to overexploitation of aquifers and water transfer between basins (Arriaga et al., 2000). According to the National Water Commission Atlas (CONAGUA, 2012), 101 of the 282 most important aquifers are currently overexploited, mainly because of excessive water extraction for agricultural irrigation. These overexploited aquifers provide 49% of subterranean water. The most serious environmental impacts include droughts in semi-arid areas that reduce flow and its timing, saline intrusion into aquifers, and wetlands ecosystem deterioration (Ávila et al., 2005).

Continuous groundwater pumping irreversibly affects natural water discharge flowing into aquatic ecosystems and riparian areas, even those that are far from mining areas. There are several cases in Mexico where the loss of fresh water that previously came from groundwater threatens the ecosystem. Such is the case of wetlands in Xochimilco, springs high Lerma and Aguascalientes, several major lakes in central Mexico (Chapala, Cuitzeo and Patzcuaro) or wildlife protected area Cuatrociénegas, among many others (Carabias et al., 2010).

In El Estor, a wetland area in Guatemala, only small wetland remnants remain; most wetlands in the area have been transformed to large-scale oil palm, sugar cane, and other crops, displacing communities and causing land conflicts among other problems (Guatemala Ramsar National Report, 2015).

The Honduras Wetland Inventory (SERNA, 2009) notes that the most affected and currently endangered systems in Honduras are humid forests and the freshwater systems within them; due to replacement with monocultures like oil palm and banana or urban lands. Honduras has implemented agreements of understanding with the private sector to carry out international certification and develop programs of good practices considering the policy and strategy of cleaner production for oil palm because it is affecting large areas of wetlands in the country. On the other hand, regulations including subsidies and incentives promoting monocultures in protected areas are under review that will, in some cases, allow for excessive development within these areas (Honduras Ramsar National Report, 2015)

Mangroves in Mesoamerica are also threatened by deforestation and aquaculture. Mexico has 5.4% of the global extent of mangroves (Giri et al., 2011), but many of those forests are being replaced with shrimp farms, agro-industrial plantations, or tourism enterprises. The threats to mangroves are similar along the Nicaraguan Pacific coast, which is unique as it marks the transition from dry to moist. The total destruction of the Estero Real mangrove in the Fonseca Gulf (between Nicaragua and El Salvador) is a clear example of the impact of uncontrolled shrimp-farm development in the region.

Caribbean

Humid and dry tropical forests are increasing overall across the Caribbean as agriculture has declined. In Puerto Rico and the Lesser Antilles, forest cover has been increasing since the 1950s (Helmer et al., 2008a,b), starting with emigration to more developed countries after the Second World War and continuing with emigration from rural to urban areas as local economies shifted from agriculture to industry and services. This shift is largely the result of sugar cane cultivation becoming less profitable due to the rise of mechanized sugar cane cultivation in South America and cessation of European price supports for banana cultivation in the Lesser Antilles (Helmer et al., 2008b; Walters, 2016). In a subset of four islands of the Lesser Antilles, cultivated land area declined 60-100% from 1950-2000, while forest cover increased 50-950% and urban land areas increased 90 to 2400% (Helmer et al., 2008b). Forest recovery will likely continue on islands like St. Kitts, Barbados, and Trinidad, where local government subsidies for sugar cane cultivation stopped only in the last decade (Helmer et al., 2008a, b; Helmer et al., 2012; Walters, 2016).

Forest recovery is most extensive in the least accessible places: at higher elevations, further from roads and urban centers, and in protected areas (Helmer et al., 2008a; Chai et al., 2009; Newman et al., 2014a). Deforestation and forest fragmentation continue in some places, including for small-scale agriculture where there is underemployment, when coffee prices are high, or in protected areas where protection is not enforced (Chai et al., 2009; Newman et al., 2014 a, b). Haiti, the poorest country in the Caribbean, lost forest cover from 2001-2010 (Alvarez-Berrios et al., 2013).

In the Caribbean, expansion of tourism and urbanization drive land-cover change rather than agriculture and cattle ranching expansion. The attraction of Caribbean islands for the development of exclusive resorts and golf courses targeted at the North American and European markets drives this land-cover change. Such tourism development plus urbanization often most severely impact tropical dry forests on Caribbean islands, because these forests are located at lower elevations and in coastal areas (Helmer et al., 2008b; Portillo-Quintero & Sanchez-Azofeifa, 2010; van Andel et al., 2016).

Development also affects water quality in freshwater and coastal systems (see 4.4.2). In the Lesser Antilles, much of the urban and residential development is for tourism and for former emigrants returning to retire (Walters, 2016). Mangrove area has declined in the Caribbean from 1980-2010 (Angelelli & Saffache, 2013), and mangrove forests continue to undergo clearing for land development (Schleupner, 2008); although, mangroves have recovered in some places where they were previously cleared for agriculture (Chinea & Agosto, 2007). Cuba alone has 3.1% of the global extension of mangroves (Giri et al., 2011).

Over 180 million people live in or travel to coastal areas of the Caribbean Sea and Gulf of Mexico annually, not counting USA coastal areas. Urban habitats have been changing rapidly in the Caribbean, with unforeseen consequences on the quality of life. An important issue has been the rapid spread of diseases, like those borne by mosquito vectors. For example, in the municipality of San Juan, Puerto Rico, the incidence of dengue fever has increased along with sea surface temperatures and sea level, as more areas for breeding become available along the shoreline and because of increasing rainfall (Mendez-Lazaro et al., 2014).

Caribbean marine ecosystems are among the most severely impacted globally (Halpern et al., 2007), mainly due to impacts on coastal systems: mangroves, coral reefs, seagrass beds and beaches (see also section 4.4.2). Live coral cover declined by 80% in 25 years in the wider Caribbean to 2001 (Gardner et al., 2003), and further declined following mass coral bleaching in 2005 (Wilkinson & Souter, 2008).

South America

Net forest loss from 2010 to 2015 in South America was dominated by forest loss in Brazil (984,000 hectares per year) and, to a lesser extent, Paraguay (325, 000 hectares per year), Argentina (297,000 hectares per year), Bolivia (289,000 hectares per year) and Peru (187,000 hectares per year) (Keenan et al., 2015). Despite the net loss of forest in South America, there has been a decline in the net rate of forest loss in some countries of the Americas (for example, in Brazil, the net loss rate between 2010 and 2015 was only 40% of that in the 1990s) and forest area increased in other countries in the last five years (for example, in Chile partly due to an increase in planted forest areas) (Keenan et al., 2015).

Deforestation and degradation of tropical rainforest are important global issues due to their role in carbon emissions, biodiversity loss, and reduction of other ecosystem services (Foley et al., 2007). Of global gross forest cover loss from 2000 to 2012, 32% occurred within tropical rainforests (Hansen et al., 2013). Almost half of rainforest loss was found in South America, primarily in the Amazon basin. Large-scale (e.g. cattle ranching) and small-scale farming were historically the most significant drivers of deforestation in the Amazon. These farming activities resulted from favorable incentives received by cattle ranchers in the 1960s–1980s. More recently, the establishment of soy farming has become a land-demanding economic activity (Kirby et al., 2006; Rudel et al., 2009). Deforestation influences Amazonian fire regimes because it results in increased sources of ignition, increased forest edge lengths, and alterations of regional climates (Alencar et al., 2015). Droughts linked to the El Niño and human-related activities were associated with large forest fires (Alencar et al., 2006; Morton et al., 2013). If climate change and increased forest degradation continue, fires may burn more frequently and expand to larger areas, perhaps including landscapes that otherwise are fire resistant (Alencar et al., 2015).

Together with lowland tropical forests, mountain areas represent an important percentage of South America (Armenteras et al., 2011). Andean forests are particularly susceptible and highly vulnerable to climate change because of their location on steep slopes and because of their altitudinal and climatic gradients (Karmalkar et al., 2008). In addition to climate change, tropical mountains are subject to high pressure from other natural and anthropogenic drivers of change like land use and land cover change, soil erosion, landslides and habitat destruction (Achard et al., 2002; Bush et al., 2004; Grau & Aide, 2008).

Together with Mexico, Brazil and Bolivia harbor the largest and best-preserved tropical dry forest fragments. The Chiquitano dry forests of Bolivia and Brazil alone extend over 142,941 km^2 (27.5% of total dry forest area in the region) (Portillo-Quintero & Sánchez-Azofeifa, 2010). Of the 23,000 km^2 of dry forest under legal protection, 15,000 km^2 are in Bolivia and Brazil. In fact, Bolivia protects 10,609 km^2 of dry forests, including 7,600 km^2 in a single park. In other countries, like Ecuador and Peru, however, low extent and high fragmentation of dry forests were observed.

Woodlands and savannas in South America are also under strong conversion rates related to the expansion of soybean and pasture (Barona et al., 2010). The Brazilian Cerrado is the second largest biome in South America and is considered a biodiversity hotspot. By 2010, approximately 50% of the original vegetative cover of the Brazilian Cerrado has been converted. Land use changes in the Cerrado, often coupled with increased fire frequency and invasion of exotic species, have generated profound changes in the vegetation structure and functioning of these ecosystems (Bustamante et al., 2012). Alterations in land cover from natural to rural and urban are also changing stream water chemistry in the Cerrado (Silva et al., 2011).

Fire is an important factor in maintaining grassland ecosystems. It prevents woody encroachment, removes dead herbaceous material, and recycles nutrients. Without fire, organic matter and litter would accumulate and tree densities would increase, leading eventually to forested areas. The timing, frequency, and intensity of fires determine specific effects of these events on the functioning of grassland ecosystems. Indigenous people in the Cerrado region have been using fire for multiple purposes (¡Error! No se encuentra el origen de la referencia. and Box 4.10).

Table 4.12. The different burning regimes used by the Krahˆo. Source: Mistry *et al.* (2005).

Burning regimes for different purposes	
Protection of roça (swidden plots)	Early dry season, around April/May
Protection of certain fruiting trees	Early dry season, around April/May
Hunting	April is perceived as the best time—small patches of Cerrado are burnt over a number of days during a hunting trip
Protection of *carrasco*	Burnt April/May every 5–6 years
Livestock	grazing Pasture burnt in mid-May—small areas burnt each year
Protection of areas of *Cerrado* from later, more intense fires	Early to mid dry season
Clearing and preparing land for planting	Roças are burnt at the very end of August or in September
Honey extraction	September and October
Keep clean and increase visibility	Throughout the dry season—fires are set when walking to villages, hunting and travelling to roças
Eliminate pests	Throughout dry season
Outsider fires	Occur throughout dry season

Box 4.10. Traditional fire management in the South America

Traditional fire knowledge is as fire-related knowledge, beliefs, and practices that have been developed and applied on specific landscapes for specific purposes by long time inhabitants (Huffman, 2013). Across the Americas indigenous people have managed fire for different purposes. The articulation of traditional and scientific knowledge can be a valuable strategy for the formulation of environmental policies for effective fire management.

Indigenous peoples have been using fire in the Cerrado (savannas) of Brazil as a form of management for thousands of years. Mistry et al. (2005) studied the traditional use of fire as a management tool by the Krahˆo indigenous group living in the northeastern region of Tocantins state, Brazil. The results indicate that the Krahˆo burn for a variety of reasons throughout the dry season, thereby producing a mosaic of burned and unburned patches in the landscape Table 4.12. Similarly, in Canaima National Park, Venezuela, a protected area inhabited by the Pemón people, ecological studies have revealed that the creation of a mosaic of patches with different fire histories could be used to create firebreaks that reduce the risk of the wildfires that threaten the vulnerable and diverse savanna-forest transition areas (Bilbao et al., 2010). In the Amazon region, particularly along large and small rivers, are numerous patches of Amazonian dark earth (Junqueira et al., 2010). These are anthropogenic soils associated with archaeological sites, created mostly between 1000 BC and the European conquest around 500 years ago and managed with the use of fire (Rebellato et al., 2009). Pre-conquest Amazonian peoples used fire for most of their landscape management. Small areas were weeded with wooden digging sticks and wooden machetes, while occasional small trees were cut with stone axes and burned well before being completely dry and/or with low oxygen availability, leaving large amounts of charcoal instead of easily eroded ash (Denevan, 2004). The combination of fire management and plant cultures improved soil fertility and once a plot was abandoned growth of secondary forests was rapid (Junqueira et al., 2010).

Similarly, vegetation cover loss in the dry Chaco from 2002 to 2006 was associated to the rapid expansion soybean and planted pastures (Clark et al., 2010). During this period a net loss of 6.9 million hectares of closed-canopy (≥80% cover) was detected in the dry Chaco ecoregion. Some of the loss of woody vegetation can be attributed to forest degradation, where forests have trees and shrubs removed as an intermediate step to agriculture or pastures (Clark et al., 2010).

Change in South American grasslands (distinguished from grasslands found in dryland regions that generally did not evolve with large mammalian herds) has been brought about primarily by conversion of these ecosystems to agriculture. The Río de la Plata grasslands are one of the largest temperate grasslands regions of the world, covering nearly 700,000 km 2 of eastern Argentina, southern Brazil and Uruguay (Paruelo et al., 2007). This region plays a key role in international crop production and land use change rates in some areas and are among the highest detected nowadays. Agricultural activities have undergone important changes during the last 20 years because of technological improvements and new national and international market conditions for commodities (mainly soybean, sunflower, wheat, and maize) (Baldi & Paruelo, 2008).

Wild ungulates are also an essential component of energy and nutrient flows in grassland ecosystems that evolved with grazing. By contrast, domestic livestock generate effects that are disputed as either positive or negative, particularly in relation to different stocking densities, different grassland environments and whether the different environments evolved with large mammalian herds (Mack & Thompson, 1982). The economic and environmental sustainability of beef cattle from pasture use and preservation in specific biomes is still not well evaluated. The study of the feasibility of beef production in the pampa biome suggests it is possible to optimize low greenhouse gases emission of beef production with a significant economic return under certain feed conditions. Actually, studies suggest it is possible to obtain beef production increases without the need of new livestock areas, which can contribute to the proper use and preservation of the pampa biome (Ruviaro et al., 2016, see also Modernel et al., 2016).

Afforestation of some of the most productive native grasslands of the region is currently undergoing, and might be further promoted by carbon markets (Paruelo et al., 2007) posing a new threat to these ecosystems. Interestingly, grasslands store approximately 34% of the global stock of carbon in terrestrial ecosystems while forests store approximately 39% and agroecosystems approximately 17%. Unlike tropical forests, most of the grassland carbon stocks are in the soil.

Drylands cover more than 50% of South America. The region possesses tropical, highland, coastal and continental drylands (Cabrera & Willink, 1980). In South America, humans have appropriated much of the Sechura Desert (Peru) for their use, and the habitat is highly fragmented (Hoekstra et al., 2010). Similarly, the Atacama Desert (Chile) has experienced moderate land appropriation for human use and moderately high habitat fragmentation (Hoekstra et al., 2010). In Patagonia, heavy sheep grazing has locally extirpated preferred forage species, thus altering plant community composition and resulting in the endangerment of 76 grass species (Cibils & Borrelli, 2005). Aside from grazing, this region has experienced a relatively low appropriation of land for human use, but has very high habitat fragmentation (Hoekstra et al., 2010). As with the other deserts, it does not have a natural fire cycle. Habitat loss in all three regions has been relatively low (0.1% for Atacama Desert, 0.5% for Sechura Desert, and 1.6% for the Patagonia steppe) (Hoekstra et al., 2010).

From 2001 to 2013, 17% of new cropland and 57% of new pastureland replaced forests throughout Latin America (Aide et al., 2013). Cropland expansion from 2001 to 2013 was less (44.27 Millions of hectares) than pastureland (96.9 Millions of hectares), but 44% of the 2013 cropland total was new cropland, versus 27% of the 2013 pastureland total, revealing higher regional expansion rates of row crop agriculture. The majority of cropland expansion was into pastureland within core agricultural regions of Argentina, Brazil, Bolivia, Paraguay, and Uruguay (Graesser et al., 2015; Volante et al., 2015). Commodity crop expansion, for both global and domestic urban markets, follows multiple land change pathways entailing direct and indirect deforestation, and results in various social and environmental impacts (Meyfroidt et al., 2014).

Forested wetlands in the western Amazon, have declined only moderately in area in recent years but local deforestation is more intense in the eastern Amazon. Habitat loss in that region is mostly concentrated in the vicinity of very large cities and in the Amazon estuary (Magalhães et al., 2015). The anthropization of these wetlands involves the forest cover removal, or alternatively, sudden changes in forest composition (Freitas et al., 2015). Natural wetland habitats are continually transformed into croplands and pastures (Junk et al., 2014).

In recent years many new large dams have been planned for the Amazon and its connection to the Andes

(Finer & Jenkins, 2012; Fearnside, 2013), causing deforestation and habitat loss (mainly riverine habitats, forming wetland patches along the river side) as main impacts (among others) (Lima et al., 2014, Cunha & Ferreira, 2012; Ferreira et al., 2013). Further, dam construction comes with huge social and economic costs involved (Fearnside 2005 and 2015). About 60% of the rural population lives inside várzeas (basin), and all major large cities are inside or on the border of flooded environments. Most timber and a significant part of the beef, fruits and vegetables consumed in urban areas are produced in these wetlands. Additionally, most of the fish consumed come from the white-water rivers and their floodplains (Junk et al., 2012). Wetlands also provide other benefits to people (Castello et al., 2013b, Junk et al., 2014), particularly because they retain nutrient rich sediment that forms new soil, control erosion, and sequester carbon dioxide.

The intense loss of natural habitats and associated biodiversity is causing the slow degradation of South American wetlands, reducing natures benefits to people by reducing the number of commercial fish species, total fish stocks, and a persistent “fishing-down” process Castello et al., 2013; Cella-Ribeiro et al., 2015), as well as the loss of carbon dioxide sinks where land-use change has been intense (Schöngart et al., 2010; Vogt et al., 2015).

Unregulated markets for timber and fish (Soares-Filho et al., 2006; Junk et al., 2007), among other natural resources harvested from the Amazonian wetlands, are the main source of illegal pressure on the extraction rates of those resources. Rural-urban migration in the Amazon, closely related to wetlands, has contributed to urban degradation, and also puts pressure on rural exploitation, affecting forest extent, since important rural patterns of consumption are maintained (Padoch et al., 2008).

The marine areas of South America include almost 30,000 km of coastline and encompass three different oceanic domains—the Caribbean, the Pacific, and the Atlantic (latitude range from 12ºN to 55ºS) (Miloslavich et al., 2011). Habitat transformation (for infrastructure expansion, aquaculture, agriculture, etc.), and sewage and garbage disposal are among the most recurrent problems in South America coastal zones. As such, these areas undergo fast and frequently drastic transformation. When compared to other tropical regions like Southeast Asia, the importance of aquaculture in South America is relatively small. Nonetheless its importance is growing in countries like Ecuador, where a significant shrimp mariculture industry has developed mostly in mangrove converted areas and salt ponds and in Peru and Chile (Humbolt Current region) with the cultivation of introduced salmonid species (Campuzano et al., 2013). In the tropical west Atlantic major threats are industrial (trawling) and artisanal (line and longline) fishing, urban development, agriculture development, dredging and flow navigation, water pollution (runoff from smaller rivers as in terms of volume the Orinoco and Amazon discharge is relatively pristine), mangrove deforestation, activities related to oil and gas exploitation, port activities, and maritime shipping (Klein et al., 2009).

Mangroves in South America correspond to 11% of the global mangrove extent (Giri et al., 2011). In the Brazilian shelf, mangrove ecosystems cover 16 of the 17 Brazilian coastal States, representing 85% of the coastline (about 7,300 km), and the extent of mangroves along the Brazilian coastline from east of the Amazon River mouth (Pará) to the Bay of São José (Maranhão) constitutes the largest continuous belt globally (Nascimento et al., 2013). Although almost 83% of mangrove areas are protected, human settlements along the coast have dramatically increased, impacting mangroves by diverting freshwater flows and degrading water quality. Mangroves also undergo salt extraction and conversion to agriculture, aquaculture (mainly shrimp farms), or built-up lands, all of which contribute to mangrove degradation and deforestation (Magris & Barreto, 2010). Despite its value, the mangrove ecosystem is one of the most threatened on the planet. Mangroves are being destroyed at rates three to five times greater than average rates of forest loss and over a quarter of the original mangrove cover has already disappeared; this destruction is driven by land conversion for aquaculture and agriculture, coastal development, pollution and overexploitation of mangrove resources. As mangroves become smaller and more fragmented, important ecosystem goods and services will be diminished or lost. The consequences of further mangrove degradation will be particularly severe for the well-being of coastal communities in developing countries, especially where people rely heavily on mangrove goods and services for their daily subsistence and livelihoods (Valiela et al., 2001; Duke et al., 2007; UNEP, 2014).

South America's west coast is home to approximately 40 million people. In Chile, three quarters of the population lives and works along a 500 kilometer stretch of coastline between Valparaiso and Concepcion, representing 15% of the country's land area. In the east coast, over 15 million people live in the Buenos Aires-La Plata-Montevideo coastal region. The coastal area between Sao Paulo and Rio de Janeiro, Brazil, hosts over 30 million people. Each of these areas continues to grow in population. The marine and inland waters are used for food production, transportation, tourism, and water supply and are important for the economic and social vitality of these communities. These aquatic ecosystems are exposed to resource use and extraction by a range of activities, from oil and gas to fisheries, from urbanization to agriculture. These activities lead to sediment, nutrient, or other pollutant inputs from the watershed (section 4.4). Many coastal, estuarine, and fresh water systems in the region have in the past seen intense outbreaks of cholera and other water-borne diseases, dengue fever and other mosquito-borne diseases, as well as an increase in the occurrence of harmful algal blooms. Some of these are due to population growth and eutrophication, but climate variability complicates the situation.

An important factor that affects the coasts and shelf environments is riverine discharge. Discharge affects the amount of sediment and nutrients that may be delivered to the coastal zone, and this in part depends on uses of the land in the watershed. As weather patterns of the future are still uncertain, the impact on global coastal systems is also a matter of speculation. Many rivers are intervened by damming, and many have different nutrient inputs due to point and non-point sources of nutrients and pollutants (section 4.4.2)

4.4.2 Pollution and related changes in biogeochemical cycles

Nature of the driver, its recent status, and trends and what influences its intensity

In its pursuit of food, water and civilization, humanity mobilizes chemicals that impact biodiversity and NCP. Pollutants (Table 4.13) are a major driver of declinesin freshwater systems, which are now, in many cases, severely degraded (Dudgeon et al., 2006). Besides changing climate (section 4.4.3), increased concentrations of atmospheric carbon dioxide adversely impacts marine species through ocean acidification. Pollutants also affect biodiversity because their human use to increase food, energy or minerals alters air, water and soil chemistry, or disturbs watersheds, causing soil erosion and sediment movement into water bodies. Other pollutants are toxic to organisms.

Table 4.13. **Examples of ubiquitous water pollutants (A) micropollutants; (B) macropollutants and fluxes to world rivers. Source: modified from Schwarzenbach *et al.* (2006) and references therein.**

A Origin/usage	Class	Selected examples	Related problems
Industrial chemicals	Solvents	Tetrachloromethane	Drinking-water contamination
	Intermediates	Methyl-t-butylether	
	Petrochemicals	BTEX (benzene, toluene, xylene)	
Industrial products	Additives	Phthalates	
	Lubricants	PCBs (polychlorinated biphenyls)	Biomagnification, long-range transport
	Flame retardants	Polybrominated diphenylethers	
Consumer products	Detergents	Nonylphenol ethoxylates	Endocrine active transformation product
	Pharmaceuticals	Antibiotics	Bacterial resistance, nontarget effects
	Hormones	Ethinyl estradiol	Feminization of fish
	Personal-care products	Ultraviolet filters	Multitude of (partially unknown) effects
Biocides	Pesticides	Dichlorodiphenyltrichloroethane (DDT)	Toxic effects and persistent metabolites
		Atrazine	Effects on primary producers
	Nonagricultural biocides	Tributyltin	Endocrine effects
		Triclosan	Nontarget effects, persistent degradation product (methyl-triclosan)
Geogenic/natural	Heavy metals	Lead, cadmium, mercury	
	Inorganics	Arsenic, selenium, fluoride, uranium	Risks for human health
	Taste and odor	Geosmin, methylisoborneol	Drinking-water–quality problems
	Cyanotoxines	Microcystins	
	Human hormones	Estradiol	Feminization of fish
Disinfection/oxidation	Disinfection by-products	Trihalomethanes, haloacetic acids, bromate	Drinking-water–quality, human health
Transformation prods.	Metabolites from all above	Metabolites of perfluorinated compounds	Bioaccumulation despite low hydrophobicity
		Chloroacetanilide herbicide metabolites	Drinking-water–quality problems

B	**Examples of aquatic macropollutants and fluxes or mass of anthropogenic production million metric tons yr^{-1}**		
	Total inorganic nitrogen fluxes to world rivers (∼75% anthropogenic)	21	
	Total phosphorus fluxes to world rivers (60% anthropogenic)	5.6	
	Anthropogenic inputs of heavy metals Zn, Cr, Ni, Pb, Cu, Cd, Hg	0.3-1	
	Global fertilizer production (2000)	140	
	Global pesticide production	5	
	Synthetic organic chemicals production	300	
	Oil spills (average 1980-2000)	0.4	
	Plastics, Microplastics	*5-13	

*Clark *et al.* (2016)

Ocean acidification, deoxygenation and plastics pollution

As atmospheric carbon dioxide increases, mainly from fossil fuel combustion, pH and calcium carbonate saturation in ocean water decrease (Fabry et al., 2008). This is adversely impacting marine ecosystems and biota (Cooper et al., 2008; Fabry et al., 2008; Albright & Langdon, 2011; Anthony et al., 2011; Pandolfi et al., 2011; Bramanti et al., 2013; Courtney et al., 2013; Webster et al., 2013; Hall-Spencer et al., 2008). Many marine animals, like plankton, mollusks, sea stars, corals, snails and other groups, extract calcium carbonate from seawater to form their skeletal structures or shells. Ocean acidification reduces the calcium carbonate availability. The ocean is also undergoing deoxygenation. Ocean oxygen content declined 2% since 1960 and with climate change could decline an additional 1 to 7% by 2100. In the upper water column, warmer waters from global climate change drive this deoxygenation by reducing oxygen solubility; at lower depths, reduced mixing is the chief driver. Along coastlines, rivers with large nitrogen and phosphorus loads draining from fertilized agricultural watersheds, or from sewage and atmospheric nitrogen depostion, cause low oxygen levels and hypoxic “dead zones” (Diaz & Rosenberg, 2008; Rabalais et al., 2014; Schmidko et al., 2017). Hypoxic coastal waters have grown exponentially (Vaquer-Sunyer & Dwarte, 2008). The intensity and duration of hypoxia controls its impacts on biodiversity. The combination of warmer water, acidification and deoxygenation are likely interacting to negatively impact marine organisms (Bednarsk et al., 2016).

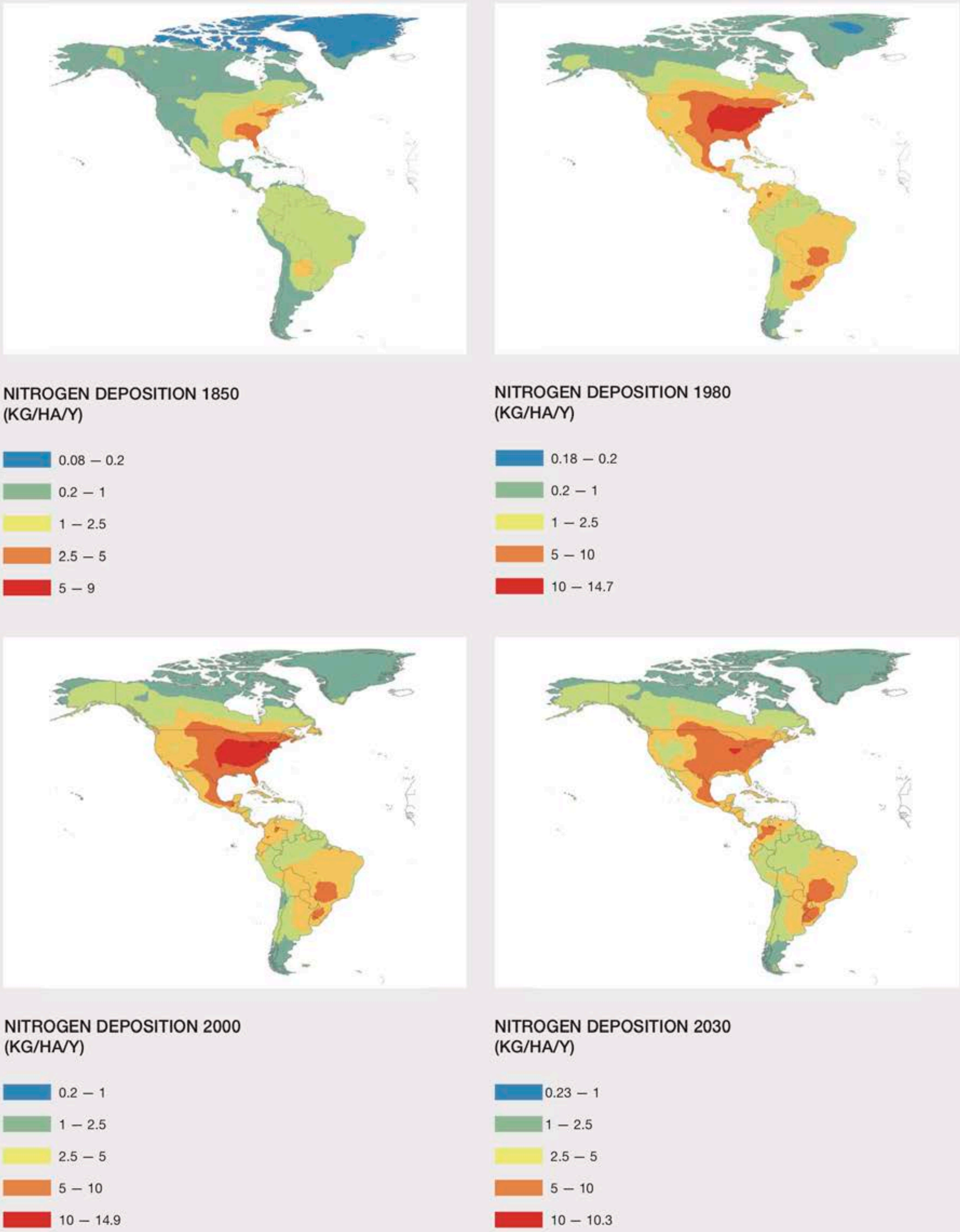
Plastic pollution enters the ocean via rivers, sewage, fishing and other sources. Plastic characteristics, like lower natural resource use and costs, and resistance to degradation, drive consumer plastics use. Although waves and sunlight break plastics to smaller pieces including microplastics (<5 mm), non-bouyant plastics take hundreds of years to degrade in ocean waters and comprise 90 to 99% of ocean plastic pollution. Plastics kill or harm biodiversity, from zooplankton, to fish, shellfish, sea turtles, seabirds and marine mammals: animals frequently consume plastics or are suffocated or maimed by them. Impacts on marine wildlife include entanglement, ingestion, and contamination to a wide variety of species. Reductions in plastics use and disposal into the oceans wouold require policy development as well as consumer-driven changes in plastics use and disposal (Wilcox et al., 2016). Many of the environmental implications of microplastics at sea are still largely unknown, however the number of marine species known to be affected by this contaminant has increased from 247 to 680 (Gall & Thompson, 2015). Microplastics have a complex effect on marine life. They adsorb legacy persistent organic pollutants and are passed up the food chain to higher trophic levels including to people, exposing humans and animals that consume marine biota to carcinogens and teratogens (toxic to embryos) (Clark et al., 2016; Worm et al., 2017). By fouling boats and fishing nets and equipment, plastic pollution imposes costs to the fishing industry and society for related cleaning and rescue (Clark et al., 2016; Kershaw et al., 2011). The top 20 countries’ mismanaged plastic waste encompass 83% of the total in 2010 with Brazil in 16th position and the USA in the 20th position in the global ranking (Jambeck et al., 2015).

Fertilization of Earth with nitrogen, phosphorus and other nutrients from human activities.

Food, fiber and energy production are changing the biogeochemical cycles of major nutrients (nitrogen, carbon, phosphorus, sulfur). The use of nitrogen, phosphate and potash fertilizer is increasing by 1.9% per year in the Americas, contributing to increasing nitrogen deposition onto ecosystems (Figure 4.5). Demand for these agrichemicals will continue to increase, mainly due to increased demand in Latin America (FAO, 2011 and 2017). Increased biologically available, reactive nitrogen (all nitrogen forms except molecular nitrogen, N_2) is the most dramatic change (Rockström et al., 2009; Jia et al., 2016). Nitrogen is central to ecosystem productivity (LeBauer & Treseder, 2008; Elser et al., 2009). In terrestrial systems, direct toxicity of nitrogen gases, ozone and aerosols, increased nitrogen availability, soil-dependent acidification, and secondary stress and disturbance, are ecosystem- and site-specific impacts that can contribute to species composition changes and reduced plant diversity (Valliere et al., 2017; Bobbink et al., 2010). Inorganic nitrogen fertilizer use releases reactive nitrogen to the atmosphere. In addition, concentrated animal feeding operations have emerged across the Americas. Animals (pigs, chickens, cows, fish and other animals) are confined, with large amounts of waste and ammonia produced. Applying this manure to agricultural fields can lead to pathogen and nutrient runoff into ground and surface waters. Increasing fossil fuel combustion, particularly coal burning for electricity, has also increased emissions of reactive nitrogen, including nitric oxide and ammonia, and sulfur dioxide. Emissions from large portions of North America have increased by more than 1,000% (van Aardenne et al., 2001).

Figure 4 5 **Total nitrogen deposition (wet and dry deposition of nitrogen oxides and reduced nitrogen) derived from the multi-model global datasets for nitrogen deposition from Lamarque *et al.* (2013).**

Data at resolution of 0.5*0.5 degrees and in units of kg N/ha/yr. 1850, 1980, 2000 and 2030 (rcp4.5). Nitrogen deposition is greatest in major agricultural regions. Source: Lamarque *et al.* (2013).



Nitrogen release can change ecosystem structure and function, affecting plant or microbial community composition, production, soil properties and susceptibility to fire or disease (Porter et al., 2013). These changes can affect recreation, drinking water quality, timber production, fisheries, wildlife viewing, climate stability, fire risk, and “non-use” values of intact, natural ecosystems (Compton et al., 2011). Runoff from agricultural fields, point sources of municipal waste (from human waste and manufacturing), and urban runoff, can transport nutrients and sediment to rivers and streams. This can increase nutrient (phosphorus, nitrogen, and carbon) concentrations and promote algal and aquatic vegetation growth causing eutrophication (Box 4.17 and Box 4.20). In aquatic eutrophication, high levels of organic matter from fertilizer and sediment run-off, and organisms decomposing it, deplete water oxygen, killing organisms including fish. It can also shift primary producer communities, alter species composition and decrease plant diversity (Box 4.17). Increased organic matter can also affect drinking water suitability and cause algal blooms that release toxins (Bushaw-Newton & Sellner, 1999; Lopez et al., 2008; Michalak, 2015; Glibert et al., 2006). Urea from fertilizer is also associated with increased paralytic shellfish poisoning along Americas coasts (Glibert et al., 2006; Glibert, 2017). These nutrient flows increase as per capita GDP, food crop and meat and milk production increase (Figure 4.6).

Figure 4 6 **Anthropogenic drivers of nutrient flows for eight world regions for 1970, 2000, and 2030 for two scenarios:**

Global Orchestration (GO) (supra-national environmental regulation) and Adapting Mosaic (AM) (localized ecosystem management). Source: Seitzinger *et al.* (2010).

AFR, Africa; SAM, South America; OCE, Oceania; SAS, South Asia; EUR, Europe; NAM, North America; AUS, Australia; NAS, North Asia

Rivers and streams naturally carry some uncontaminated sediment. However, increased land disturbance, primarily from agriculture and urbanization, can mobilize excessive amounts of fine sediment into streams. Excessive sedimentation can directly harm organisms. With mussels, for example, it buries adults and juveniles or interrupts respiration or feeding. In rivers, suspended sediments and sediment deposits may also bury eggs, displace host fish, or disrupt host fish/mussel interactions leading to declines of some species. Excessive sediment may also block light penetration into water, reducing primary production and causing the need for river channel dredging for ship traffic. Conversely, on many major rivers, dams for hydroelectric power and irrigation water have reduced river sediment loads. Lack of sediment can reduce habitat, excessively scour river channels and banks, and cause losses of coastal wetlands that depend upon a steady sediment supply (Morang et al., 2013).

Toxicants

Ecosystems throughout the world have experienced low-level exposure to many different toxicants due to human activities. Low-level exposure to toxicants may occur via air (e.g. tropospheric ozone), water (e.g.

trace metals, methyl mercury, pharmaceuticals), soil or sediments (e.g. lead, polycyclic aromatic hydrocarbons), or food (pesticides, microplastics, bioaccumulative toxics). Toxicants released to the air are disseminated the longest distances and affect the most species.

Because biota experience toxicants in combination with other stressors (water stress, altered thermal regime, habitat destruction, etc.), toxicant effects are often difficult to ascertain. Much evidence of the adverse effects of low-level toxicant exposure on biodiversity is in the literature on point sources of trace metals to aquatic habitats. We have known since the 1980's that changes in community composition occur at metal concentrations much lower than water quality criteria (Clements et al., 1988, 2000, 2013). Restoration of streams contaminated by mine drainage is often unsuccessful because the sediments have accumulated trace metals that continue low-level exposure sufficient to inhibit numerous bottom-dwelling organisms (Clements et al., 2010a, b). Metal concentrations below the chronic toxicity values on which water quality criteria are based can inhibit important ecosystem functions (e.g. photosynthesis) (Twiss et al., 2004; Sunda, 2012). These effects of low-level exposure to toxicants are consistent with the observations that abrupt changes in community composition (loss of sensitive species, loss of functional groups, decreased abundance of some species and increases in others) occur at low levels of disturbance, including low levels of pollutants (Fleeger et al., 2003; Dodds et al., 2010; King & Baker, 2010).

Atmospheric ozone occurs where emissions from fossil fuel combustion (energy utilities, industry, motor vehicle exhaust) or biomass burning interact with vapors from solvents, gasoline or vegetation. Ozone damages plant tissues, decreases plant primary production, and changes plant and insect communities (Hillstrom & Lindroth 2008; Volk et al., 2006), but its effects on biodiversity remain poorly studied.

Major sources of atmospheric mercury include fossil fuel (primarily coal) combustion (the largest source), artisanal gold mining, non-ferrous metal manufacturing, cement production, waste disposal, caustic soda production, and emissions from soils, sediment, water, and biomass burning, including re-emissions from past anthropogenic emissions (Pacnya et al., 2006; Pirrone et al., 2010). Legacy releases from commercial products and contaminated sites contribute to re-emissions (Horowitz et al., 2014; Kocman et al., 2013). In the vicinity of past or current mining, at higher latitudes, at mid latitudes with soft water ecosystems, or in regions downwind of coal fired power plants, consumers of aquatic foods may suffer high exposure to methyl mercury (Mahaffey & Mergler, 1998; Després et al., 2005; Fujimora et.al., 2012; Driscoll et al., 2007). Methyl mercury is a potent neurotoxin, and it is particularly toxic to human and other vertebrate embryos.

The discovery and development of synthetic herbicides during World War II has increased crop yields, enhanced crop quality, and reduced production and harvesting costs (Coupe et al., 2012). Possible health effects from exposure to pesticides include cancer, reproductive or nervous- system disorders, and acute toxicity. Recent studies suggest that some pesticides disrupt endocrine systems and affect reproduction by interfering with natural hormones (García et al., 2017; Gilliom et al., 2006). The amounts, types, and use of pesticides for agriculture change over time, but their worldwide use increases. Persistent organic pollutants, like organochlorine pesticides, polycyclic aromatic hydrocarbons, polychlorinated biphenyl compounds, polybrominated biphenyl ethers, and others, by being semi-volatile and resistant to degradation, are transported in the atmosphere or ocean to remote places where they can bioaccumulate and biomagnify in food webs (supplementary material: Box 4.18 and Box 4. 19). Being detectable in most global ecosystems (Bartrons et al., 2016), persistent organic pollutants should always be considered in total toxic burdens. Like methyl mercury, deposition from the atmosphere to water, soils, or sediment can be greater at colder-latitude or montane ecosystems where temperatures are colder or precipitation greater (Macdonald et al., 2000; Blackwell & Driscoll, 2015; Kirchner et al., 2009).

Agroecology is an alternative to conventional agriculture that builds on local knowledge and innovation, which could complement other agricultural approaches to contribute to sustainable intensification on farms. Organic agriculture comprises 0.8% of North American agriculture (Willer & Lernoud, 2016). In much of Latin America, agricultural fields are still managed by small farmers, despite rapid increases in industrial agriculture. Many of them practice diversified agriculture, using hand or animal power and zero or little agricultural chemicals, preserving soils and biodiversity while supplying much of the food for their countries. Networks like *Campesino a Campesino* (Farmer to Farmer) further Agroecology – the science of sustainable agriculture - by promoting exchanges of traditional knowledge and experience among farmers. Perhaps the most famous example of small-scale farmer success is Cuba. Following the Soviet Union collapse in the 1990s and the USA embargo, food production in Cuba collapsed with the loss of imported fertilizers, pesticides, tractors, parts, and petroleum. Cubans developed alternative methods of growing food. Sustainable agriculture, organic farming, urban gardens, smaller farms, animal traction, and biological pest control all became part of Cuban agriculture. They were so successful that from 1996 to 2005 Cuba sustained a 4.2% growth in per capita food production. In southern Brazil in 2008 - 2009, conventional maize farmers lost 50% of their crops in a severe drought, but farmers who followed agroecological systems lost just 20% of their maize. In Honduras, soil conservation practices introduced via *Campesino a Campesino* helped triple or quadruple the yields of hillside farmers. Many other examples of successful agroecology exist (Altieri et al., 2012; Altieri & Funes-Monzote, 2012).

North America

Atlantic and Pacific Ocean waters are more acidic since 1991, except for the subpolar Pacific (Lauvset et al., 2015; Ríos et al., 2015; Feeley et al., 2012). Arctic Ocean pH trends are not significant, but undersaturation with calcium minerals, colder waters that absorb more carbon dioxide, and low-alkaline freshwater inputs from rivers and melting sea ice, contribute to North American Arctic Ocean vulnerability to ocean acidification, including the Pacific Arctic, home to one of the world's largest commercial and subsistence fisheries (Steiner et al., 2014; Mathis et al., 2015). Large areas off the USA Pacific coast are now acidic enough to dissolve the shells of free-swimming snails (sea butterflies/pteropods), which are important in ocean food webs (Bednaršek et al., 2016). Cod larvae are highly sensitive to ocean acidification (Frommel et al., 2012).

In the USA ozone pollution from fossil fuel combustion increases human morbidity and mortality (Li et al., 2016). Springtime ozone levels are increasing in North America, which may in part be attributable to Asia (Cooper et al., 2010; Law, 2010). Emissions from motor vehicles and other fossil fuel combustion are large contributors to atmospheric fine particulate matter (Lee et al., 2003). Particulate matter is associated with premature mortality and lung cancer (Apte et al., 2015). In the USA increased infant mortality from respiratory complications, increasing the odds of sudden infant death syndrome by 25% in some studies (Woodruff et al., 1997; Son et al., 2017). Even where air meets USA standards, rates of low human birthweights increase with increasing air particulate matter (Ebisu & Bell, 2012; Hao et al., 2016). Regulations to reduce industrial and other particulate matter release to the atmosphere since the 1970s improved life expectancies in the USA (Pope et al., 2009).

Since nitrogen fertilizer production from atmospheric nitrogen gas began with the Haber-Bosch process in the early 1900s, inorganic nitrogen fertilizer use across the USA has increased (Erisman et al., 2008). Agricultural fertilizers, nitrogen deposition and nitrogen-fixing crops dominate reactive nitrogen sources, with limited areas driven by centralized sewage (point sources), manure application or urban run-off (Box 4.17). Ammonia emissions, mainly from fertilizer use, increased 9% in Canada from 1995-2000 (Schindler et al., 2006). Where oil is extracted from oil sands in North American prairie grasslands, nitrogen oxides and Sulfur emissions are increasing (McLinden et al., 2015). In the eastern USA, power plant upgrades through Clean Air Act regulations since the 1970s reduced Sulfur and nitrogen oxides deposition (though ammonia levels are increasing) (Li et al., 2015), reducing acidification of acid-sensitive lakes and rivers (Garmo et al., 2014). Recently low natural gas prices caused USA power plants to use less coal, reducing emissions of carbon dioxide (by ~23%), nitrogen oxides and sulfur dioxide (de Gouw et al., 2014). Natural gas is a potent greenhouse gas, however; leaks during its extraction, transportation and storage must be minimized (Howarth, 2014; Zimmerle et al., 2015).

Both nitrogen and sulfur atmospheric deposition can affect growth, species composition, biodiversity and ecosystem function in temperate and boreal forests of North America (Pardo et al., 2011). Nitrogen deposition's clearest impact on species is to reduce lichen and mycorrhizal diversity. They respond quickly to changes in nitrogen availability. Where soils lack minerals to neutralize acidic inputs, sulfur deposition has acidified soils, decreasing tree growth and health, and acidified runoff to aquatic ecosystems, affecting aquatic species. Atmospheric nitrogen and Sulfur deposition is also reducing diversity and increasing fire risk in some temperate grasslands and deserts of North America (Pardo et al., 2011), and it can alter diversity and ecosystem function in wetlands and freshwater systems that are naturally low in nitrogen. Nitrogen deposition may be responsible for declines in endangered species in some areas of the USA (Hernández et al., 2016).

In the USA from 1992 to 2011, pesticide concentrations exceeded aquatic-life benchmarks in many rivers and streams in agricultural, urban, and mixed-land use watersheds. The proportions of assessed streams with one or more pesticides that exceeded an aquatic-life benchmark were very similar between the two decades for agricultural (69% during 1992−2001 versus 61% during 2002−2011) and mixed-land-use streams (45% versus 46%). Urban streams, in contrast, increased from 53% during 1992−2011 to 90% during 2002−2011, largely because of fipronil and dichlorvos. The potential for adverse effects on aquatic life is likely greater than these results indicate, because potentially important pesticide compounds were not assessed. Widespread trends in pesticide concentrations, some downward and some upward, occurred in response to shifts in use patterns primarily driven by regulatory changes and new pesticide introductions (Stone et al., 2014).

In the USA agricultural use of glyphosate [N-(phosphonomethyl)glycine] has increased from less than 10,000 to more than 70,000 metric tons per year from 1993 to 2006 (active ingredient), primarily due to the introduction of genetically modified crops, particuallry corn and soybean, and is still increasing. In 2009, glyphosate accounted for >80 percent of all herbicide use on more than 31 million hectares of soybean (by weight of active ingredient). On 31.1 million hectares of corn, glyphosate accounted for about a third of herbicide use (Coupe & Capel, 2016). Glyphosate is also used in homes, and along rights of way. Glyphosate was considered more “environmentally benign” than herbicides it replaced because it has lower toxicity and mobility or environmental persistence. However, results from >2,000 samples across the USA indicate that glyphosate is more mobile and occurs more widely in the environment than was thought. Glyphosate and aminomethylphosphonic acid (a glyphosate degradation product) were detected in surface water, groundwater, rainfall, soil water, and soil, at concentrations from <0.1 to >100 micrograms per liter. Most concentrations were below adverse effects criteria, however, the effects of chronic low-level exposures to mixtures of pesticides are uncertain. Studies have attributed toxic effects to surfactants or other additives to common glyphosate formulations.

New classes of pesticides have been developed and introduced and are now widely used, but have documented environmental issues such as the persistent, systemic and neurotoxic neonicotinoids and fipronil, introduced in the early 1990s. Insecticide use has been related to the disappearance of honey bees and other insects and insect eating birds. Neonicotinoids and fipronil are found in nectar and pollen of treated crops such as maize, oilseed rape and sunflower and also in flowers of wild plants growing in farmland. They have also been detected at much higher concentrations in guttation drops exuded by many crops (van Lexmond et al., 2015).

The Laurentian Great Lakes and Greenland illustrate aspects of persistent organic pollutants in North America (Box 4.18 and Box 4.19). Persistent organic pollutants concentrations in air and fish samples in the North American Great Lakes and in some Arctic Ocean biota have slowly declined in recent decades. Polycyclic aromatic hydrocarbons decreases are from improved emissions controls (Carlson et al., 2010; Venier & Hites, 2010). Since their ban, levels of polybrominated biphenyl ethers, used as fire retardants, have declined in fish, bivalves and bird eggs in San Francisco Bay (Sutton et al., 2014). Persistent organic pollutants persist, however, and new ones are emerging. Across North America, polychlorinated biphenyls in air samples increase along a remote-rural-urban gradient. Lighter congeners are more common at higher latitudes. Polychlorinated biphenyls loadings have not declined in the Canadian Arctic, as heavier polychlorinated biphenyls are moving northwards more slowly. For polybrominated diphenyl ethers, and other emerging persistent organic pollutants, few trends have emerged (Shen et al., 2006; Braune et al., 2005; Macdonald et al., 2000).

In North America, fish mercury levels, even in remote places, are often unsafe for regular consumption by humans and wildlife in North America (Driscoll et al., 2007). Decreased reproduction in common loons, which are fish-eating birds, is correlated with female tissue mercury levels (Evers et al., 2008). In contaminated areas where fish consumption is high, human populations are at risk (Mahaffey & Mergler, 1998; Cole et al., 2004). Industrialization increased atmospheric mercury loads to remote northern lakes in North America (Swain et al., 1992; Driscoll et al., 2007; Fitzgerald et al., 1998; Durnford et al., 2010). Decreases in USA coal combustion, and environmental regulations, have reduced mercury loads to the eastern and midwestern USA have decreased, reducing mercury levels in the environment and fish (Engstrom & Swain, 1997; Evers et al., 2007; Munthe et al., 2007; Cross et al., 2015). However, the decrease in atmospheric mercury deposition in the USA has slowed, particularly in the western and central USA, which is attributed to mercury deposition from elsewhere, possibly China (Weiss-Penzias et al., 2015). In Arctic North America, mercury levels in seabird eggs and feathers, marine mammals and lake sediments are increasing (Braune et al., 2005). Emissions from Asia account for one-third of atmospheric mercury there (Durnford et al., 2010). Total mercury emissions from China increased by about 3% per year from 1995 to 2003, mostly from increasing coal burning and non-ferrous metal smelting (Wu et al., 2006).

The mercury burden in the Arctic marine food web is now 92% from man-made sources (Dietz et al., 2009), increasing an order-of-magnitude since the industrial revolution and accelerating in the 20th century. It may now cause subtle neurological or other toxic effects in many fish-eating Arctic wildlife, including Arctic toothed whales, polar bears, pilot whales, hooded seal, some bird species and landlocked Arctic char (Dietz et al., 2009). The effects of multiple pollutants, including persistent organic pollutants and mercury, are a concern among Arctic indigenous groups that frequently consume fish, marine mammals or sea bird eggs, particularly where local persistent organic pollutants sources add to background atmospheric burdens (Burger et al., 2007; Hardell et al., 2010; Hoover et al., 2012; Byrne et al., 2015). Lead contamination has also reached the Arctic from coal combustion (McConnell & Edwards, 2008). Després et al. (2005) detected correlations among tremor amplitude or other neuromotor effects and blood mercury or lead, in Inuit children in Canada. Although fish consumption increases human blood lipids that reduce cardiovascular risk and increase cognition, mercury exposure diminishes these advantages and increases cardiovascular disease indicators (Virtanen et al., 2005; Oken et al., 2005; Guallar et al., 2002).

Pollution from past and ongoing coal mining, hard-rock mining, and metal-ore smelting, expose humans, fish and wildlife to toxicants (e.g. toxic metals and selenium) across North America; thousands of mines are abandoned, and bankrupcies of mining companies are common, leaving neither public nor private funds available to to mitigate or restore these sites and allowing toxic releases and exposures to continue (Woody et al., 2010; Palmer et al., 2010; Lewis et al., 2017; Gorokhovich et al., 2003; Clements et al., 2000; Maret & MacCoy, 2002; Maret et al., 2003; Dudka & Adriano, 1997; Lovingood et al., 2004; Surber & Simonton, 2017; Hughes et al., 2016). Near past lead mining and smelting operations, ground-feeding songbirds are exposed to lead at toxic concentrations (Beyer et al., 2013). The costs to contain pollution from hard rock mining sites in the USA have spiraled upwards from tens of billions of dollars in 1993 (Lyon et al., 1993) to $75 to $240 billion today (Hughes et al., 2016).

Mesoamerica

Basin-wide acidification is increasing in oceans surrounding Mesoamerica, with pH decreasing from 1991-2011 (Lauvset et al., 2015; Bates et al., 2014). If increases in atmospheric carbon dioxide continue, many Pacific coral reef systems may no longer be viable (Feely et al., 2012). As for nitrogen deposition, studies in Mesoamerica suggest it could affect tropical forest composition by increasing soil nitrate levels that could then alter the competitive ability of nitrogen-fixing legumes or alter soil cation exchange capacity, making nutrients like calcium or potassium scarcer (Sayer et al., 2012; Hietz et al., 2011).

There are no systematic studies of agricultural chemicals in the Mesoamerica, but it appears that pesticides are frequently found in the environment. For example, glyphosate is the most commonly used pesticide in Mexico, and it was detected in water from all 23 locations sampled in one study, including protected and agricultural areas, and was higher during the dry season (up to 36.7 ug/L) (Ruiz-Toledo et al., 2014).

Pesticide use in Costa Rica more than quadrupled from 1977 to 2006, from approximately 2,650 metric tons of active ingredient to 11,600. In a study from late 2005 to 2006, pesticides were measured invarious media throughout Costa Rica (Shunthirasingham et al., 2011). Because of the variety of crops grown in Costa Rica (coffee, bananas, rice, and sugar cane) many different pesticides are used and were detected in this program, including some from fog and air samples in remote areas.

In Mesoamerica, past rather than current use appears to drive organochlorine pesticides contamination. A Costa Rican location with limited past organochlorine pesticides use has low air and soil organochlorine pesticides levels (Daly et al., 2007; UNEP, 2009). Air and soil from four Mesoamerican sites had low polychlorinated biphenyls and polybrominated diphenyl ethers levels (Shen et al., 2006), but in Mexican communities where past agricultural and antimalarial DDT (dichlorodiphenyltrichloroethane) use was high, human exposure to DDT components and dichlorodiphenyldichloroethylene is high. Children had polychlorinated biphenyls in their blood. Risk assessments should consider multiple persistent organic pollutants exposures. Metal mining concessions cover 28% of Mexico and 8% of Mexico's protected land (Armendariz-Villegas et al., 2015). Limited studies suggest that mercury levels are not elevated in sharks and rays; freshwater and marine forage fish for migratory aquatic raptors; or Pacific coastal water and sediment (Sandoval-Herrera et al., 2016; Gutierrez-Galindo et al., 2007; Elliot et al., 2015). Soils at former mining sites in Mexico have high mercury levels. (Santos-Santos et al., 2006). Though mercury may be stable in some soils (Gavilán-Garía et al., 2008), it is most toxic when methylated in wet environments, warranting surveys of mercury contamination in nearby waters. Artisanal mining still releases mercury to aquatic environments in Mesoamerica.

Caribbean

Worth almost $2 billion in 2003, the annual net benefit from Caribbean island coral reefs, excluding USA reefs, was more than the GDP of some eastern Caribbean island nations. The difference between the income they generate and their maintenance cost was almost $50 billion (Cesar et al., 2003). Forest cover increases on Caribbean islands (section 4.4.1) should reduce sedimentation to coral reefs, but concurrent urbanization could offset those benefits (Ramos-Scharron et al., 2015). Ocean acidification, pollution from human sewage, other nutrient pollution sources, sedimentation and temperature increases all contribute to Caribbean coral reef declines (Box 4.11). In addition, decreases in aragonite (calcium carbonate) saturation levels across the region (Figure 4.8) (Gledhill et al., 2008) due to acidification damages coral reef structure (Webster et al., 2013).

Few studies examine nutrient and sediment in Caribbean rivers and streams, but Puerto Rico provides an example. Beginning in the 1800s, land clearing for agriculture and urban development increased sediment and nutrient fluxes to coral reefs. A study examining sediment flux from different land uses (forest, pasture, cropland, and urban) showed that the sediment flux was higher on disturbed land and depended on the storm hydrograph, previous storms, location in the watershed, and underlying geology (Gellis, 2013). Despite much reforestation since the mid-1940s, sediment transported to river valleys from previous agriculture is still being transported through river systems. Nitrogen and phosphorous concentrations in river waters are within regulatory limits but up to 10 times higer than estimated pre-settlement levels, negatively affecting coral reefs, especially near shores. Nitrogen deposition in in Puerto Rico was associated with more soil nitrates (Cusak et al., 2015). Other anthropogenic sources of nitrogen to Caribbean ecosystems come from reforestation with molecular nitrogen-fixing trees, including exotic species (Erickson et al., 2015).

Box 4.11. Regional flattening of Caribbean Sea coral reefs

All four subregions of the Americas border the Caribbean Sea. Caribbean coral reefs have undergone a region-wide "flattening", in which an objective measure of their structural complexity, their "rugosity", which is directly related to their species diversity (Newman et al., 2015) greatly decreased from 1969 to 2008 (Álvarez-Filip et al., 2009) (see Figure 4.7). Caribbean reefs are among the marine ecosystems most impacted by humans (Halpern, 2009). Globally, Caribbean coral reefs have the most critically endangered species as a proportion of total species (Carpenter et al., 2008). Models suggest that ocean acidification and warming alone are enough to cause widespread coral mortality and reduced growth (Anthony et al., 2011). Further, overfishing that reduces populations of the fish that graze sponges or algae can also degrade Caribbean reefs (Anthony et al., 2011; Loh et al., 2015), and these same models suggest overfishing of the fish that eat algae or elevated nutrient levels will lessen coral reef resilience to ocean acidification or warming (Anthony et al., 2011). Caribbean coral reefs are subject to a variety of other stressors that reduce reef resistance to acidification (Anthony et al., 2011; Woodridge & Done, 2009).

Pollution sources include sedimentation, which represents a severe disturbance (Fabricius, 2005), and nutrient-laden runoff including sewage. Reefs are exposed to elevated nitrogen from runoff and discharges off the coast of Mexico when tourist numbers are higher (Sanchez et al., 2013). In experiments, nitrogen enrichment decreases calcification rates including for at least two dominant reef-building Caribbean corals species, and likely contributes to coral overgrowth by algae (Marubini & Davies, 1996; Fabricius, 2005). Various diseases are also devastating Caribbean reefs (Sutherland et al., 2004; Carpenter et al., 2008), including one that rapidly spreads and kills a primary reef building species in the Caribbean, Elkhorn coral (*Acropora palmata*) and that is linked to human sewage (Patterson et al., 2002; Sutherland et al., 2010).

Acidification in the greater Caribbean Sea is demonstrated by a clear long-term decrease in pH and an increase in surface water dissolved carbon dioxide between 1996 and 2016 (see Bates et al., 2014; Astor et al., 2013) and a strong decrease in aragonite (calcium carbonate) saturation levels across the region (Figure 4.8) (Gledhill et al., 2008). Decreases in aragonite saturation due to acidification can inhibit maintenance and recovery of coral reef structure (Webster et al., 2013), and for coral reefs to remain in coastal Caribbean areas, they will have to recover from local- to large-scale physical and other disturbances like those from hurricanes or coral bleaching (Goreau, 1992; Carpenter et al., 2008), both of which can kill coral, and from ocean warming (Yee et al., 2008; Pandolfi et al., 2011), which leads to bleaching. Increasing atmospheric carbon dioxide depresses metabolism, settlement and growth of larvae of the important Caribbean reef-building species *Porites astreoides* (mustard hill coral) (Albright & Langdon, 2011). Related *Porites sp.* of the Indo-Pacific show declining calcification rates over the past 16 years, and Cooper et al. (2008) attribute this change to ocean acidification. Experiments with other Caribbean species, like the reef urchin (*Echinometra viridis*), also show impaired calcification of Caribbean reef species (Courtney et al., 2013).

Figure 4 7 **Changes in reef rugosity on reefs across the Caribbean from 1969 to 2008 with examples of four different values of rugosity index of architectural complexity on Caribbean reefs.**

Black dots at the top of the figure indicate the significant breakpoint in 1985 and 1998 (+1 s.e.) for the segmented regression. Source: Álvarez-Filip *et al.* (2009). Photos courtesy of Lorenzo Álvarez-Filip.

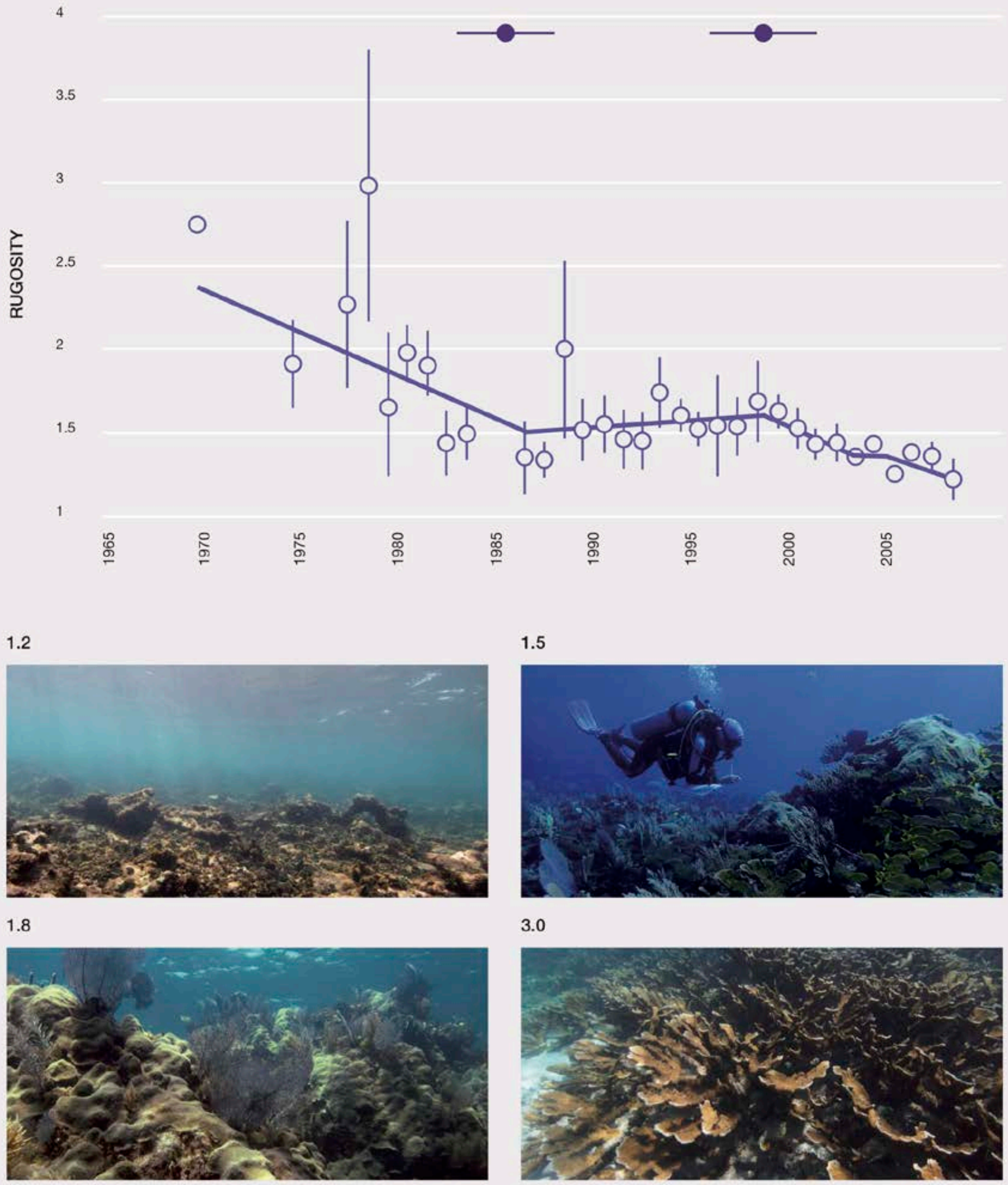
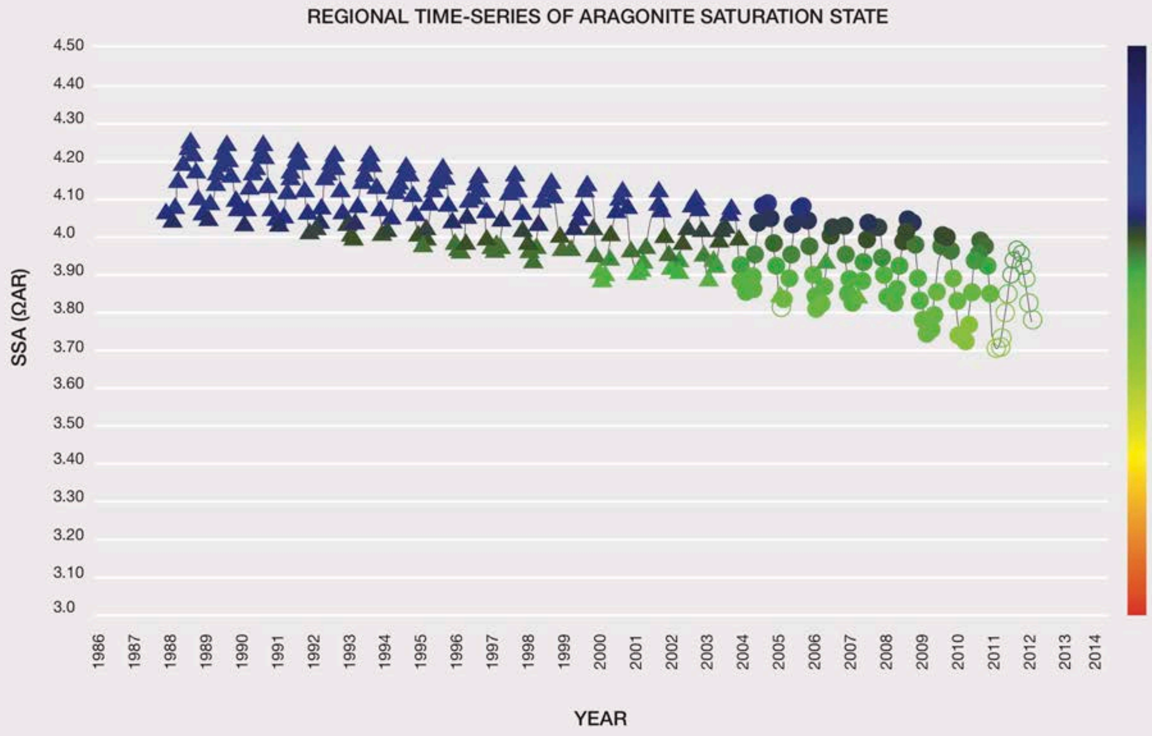


Figure 4 8 Trends in mean monthly sea-surface aragonite saturation (SSA) (aragonite is a form of CaCO3) in the Caribbean sea since the late 1980s.

Declines in SSA are related to declines in coral reef accretion. Sources: Gledhill *et al.* (2008), figure updated data downloaded from: https://coralreefwatch.noaa.gov/satelli te/oa/saturationState_GCR.php.



Caribbean island cloud forests and biota can have high mercury levels (Townsend et al., 2013), suggesting that global atmospheric mercury burdens are affecting them, given that these forests are cooler, wetter and intercept fog. Caribbean cloud forest soils are often waterlogged (Silver et al., 1999), which could spur mercury methylation. As in Mesoamerica, past use of legacy organochlorine pesticides is associated with high concentrations in streams, coastal environments and biota. Past chlordecone use in Martinique and Guadeloupe is associated with current concentrations in freshwater and coastal ecosystems (Coat et al., 2006, 2011; Charlotte et al., 2016). Low-level chronic exposure of developing infants and infants to chlordecone negatively impacts infant cognitive and motor developments in Guadeloupe (Dallaire et al., 2012).

South America

Ocean acidification is increasing around South America; pH decreased from 1991-2011 in the southern and equatorial Atlantic and Pacific Oceans and the subpolar southern Ocean (Lauvset et al., 2015). Southern Ocean systems are highly vulnerable to ocean acidification. Colder waters hold more carbon dioxide and dissolve more calcium carbonate. Species critical to the pelagic or benthic southern Ocean food web, including Antarctic krill (*Euphausia superba*), some pteropods, and benthic marine invertebrates, could collapse from ocean acidification alone, ignoring temperature changes (Kawaguchi et al., 2013; McNeil & Matear, 2008; McClintoc et al., 2009). Experiments show that species from subtropical southern Pacific Ocean waters are vulnerable to ocean acidification (Vargas et al., 2015). Upwelling, rainfall, tides and river flows (Vargas et al., 2016; Manzello, 2010) affect seawater carbon dioxide levels, upwelling around the Galapagos Islands, cause high carbon dioxide levels and low calcium carbonate, places its waters near the distributional limits for coral reefs, making them particularly vulnerable to ocean acidification (Manzello, 2010).

The worldwide need for food and increased rainfall as led to agricultural expansion and change over recent decades in South America. Rapid adoption of genetically modified crops has occurred, particularly glyphosate tolerant soybean and corn and Bt-corn and cotton (De la Casa & Ovando, 2014; Brookes &

Barfoot, 2011). Between 1996 and 2009, the area planted to soybeans in Argentina increased by 215% (from 5.9 to 18.6 million hectares) (Lapola et al., 2014).

Agriculture has intensified over the same period, with one field producing two to three crops per year. Water-quality degradation in Brazilian rivers is directly proportional to agricultural extent in watersheds and riparian zones.

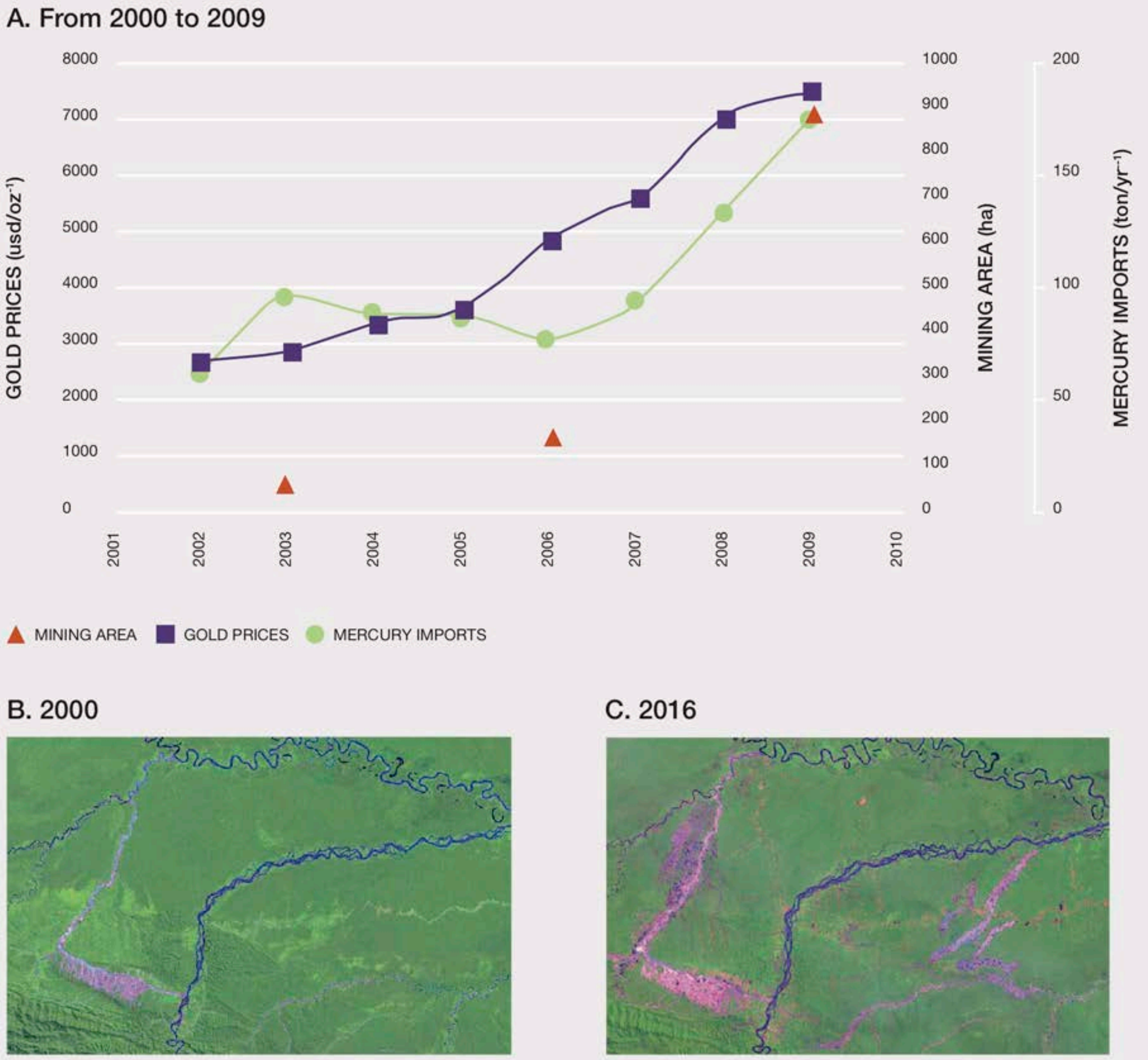
There are no systemic studies of agricultural chemicals in the South American environment, but given the large use of glyphosate on genetically modified soybean it can be assumed that conditions are similar to the USA where glyphosate can be found in every environmental compartment (Coupe et al., 2012; Battaglin et al., 2014; Rios et al., 2010).

Total dissolved nitrogen yields in major South American rivers, including the Río de la Plata and Amazon, are less than many major world rivers. Rivers with the highest total dissolved nitrogen yields in South America pass through heavily populated areas - they lack of municipal and industrial treatment plants. Rivers impacted by agriculture have lower total dissolved nitrogen yields. Water pollution in South America is dominated by municipal and industrial sewage (Bustamente et al., 2015). In all countries of the Amazon and Orinoco River basins, wetlands and major rivers show pollutant impacts on biodiversity (Crema et al., 2011; Gomez-Salazar et al., 2012; Lopes & Piedade, 2014). Where Amazonian wetlands (forested floodplains, marshes, wet meadows, peatlands, tidal wetlands, etc.) are densely populated, conversion to agriculture, accompanied by fertilizer organic matter loads, cause super or even hypereutrophic areas in the mid-lower course of the Amazonas River (Affonso et al., 2011). Increased nitrogen availability from agriculture, mining, sewage pollution, shrimp farming and solid waste disposal threaten South American mangroves (Lacerda et al., 2002; Castellanos-Gallindo et al., 2014; Rodríguez-Rodríguez et al., 2016) (supplementary material: Box 4.20**).**

Petroleum drilling is increasing in the Amazon; repeated spills contaminate water, sediment and soils with toxic hydrocarbons or metals (Frazer, 2016; Marínez et al., 2007) in many indigenous communities. This income source is also a public health concern: childhood leukemia and spontaneous abortion are higher among people living near oil drilling, and stream water exceeds allowable limits for petroleum hydrocarbons (San Sebastián & Hurtig, 2004; San Sebastián et al., 2002). Despite such concerns, little related research is available (Orta Martínez et al., 2007; Orta-Martínez & Finer, 2010), but water and sediment near oil-related activities can be highly contaminated with polycyclic aromatic hydrocarbons and mutagenic (Reátegui-Zirena et al., 2013), and drilling fluids have high toxic metal concentrations. Oil exploration is a source of spills that affect wetlands (Lopes & Piedade, 2014). In general, metal-polycyclic aromatic hydrocarbons mixtures have a more than additive toxicity effect on aquatic invertebrates (Gauthier et al., 2014). Oil and dispersants are toxic to Amazonian fish (Pinto et al., 2013). As of 2008, around180 concessions for oil exploration or extraction, involving ≥35 companies, cover much of the most species-rich part of the Amazon (Finer et al., 2008), subjecting the area to pollution and opening it to deforestation and hunting (Butt et al., 2013).

Amazonian countries are large and increasing sources of mercury emissions from artisanal gold mining (Telmer & Veiga, 2009). Mining area correlates with gold prices (Swenson et al., 2011) (Figure 4.9). Although some mercury leaches from soils (Fadini & Jardim, 2001), most mercury contamination is anthropogenic, and seasonal flooding disperses it. Higher mercury concentrations occur downstream from mining sites in fish, sediment and humans (Malm, 1998; Mol et al., 2001; Cordy et al., 2011; Fujimura et al., 2012). Its adverse effects on vertebrate embryos and the human nervous system are well known (e.g. Passos & Mergler, 2008).

Figure 4 9 **Relationships among gold prices, mercury imports to peru, and forest clearing for gold mining from 2000 to 2016 in a region of peru. Source: Swenson *et al*. (2011); USA Geological Survey.**



In South America also, higher legacy of persistent organic pollutants levels occur where past use was high. In a Patagonian watershed of Argentina, river water, sediments and wetland soils had higher polychlorinated biphenyls and organochlorine pesticides concentrations near agriculture, urban areas and hydroelectric facilities (Miglioranza et al., 2013), and raptors may have high organochlorine pesticides levels (Martínez-López et al., 2015). In coastal areas, a protected estuary receiving sediment from nearby urban and industrial areas had high polychlorinated biphenyls concentrations (Pozo et al., 2013). Like the Arctic, long-range transport of polychlorinated biphenyls is still increasing in remote mountain lakes in Chile (Pozo et al., 2007).

In air samples from the Cauca valley of Colombia, higher persistent organic pollutants compared with other places in Latin America are presumably associated with the extensive urban and agricultural areas (Álvarez et al. 2016). Sediment cores from the Santos estuary of Brazil show that polycyclic aromatic hydrocarbons increased over time with development (Martins et al., 2011).

4.4.3 Climate Change

Nature of the driver, its recent status and trends, and what influences its intensity

Climate change is defined as “Any change of climate which is attributed directly or indirectly to human activity that alters the composition of the global atmosphere greenhouse gases (carbon dioxide, methane, methane and nitrous oxide) over comparable time periods.” (IPCC, 2013).

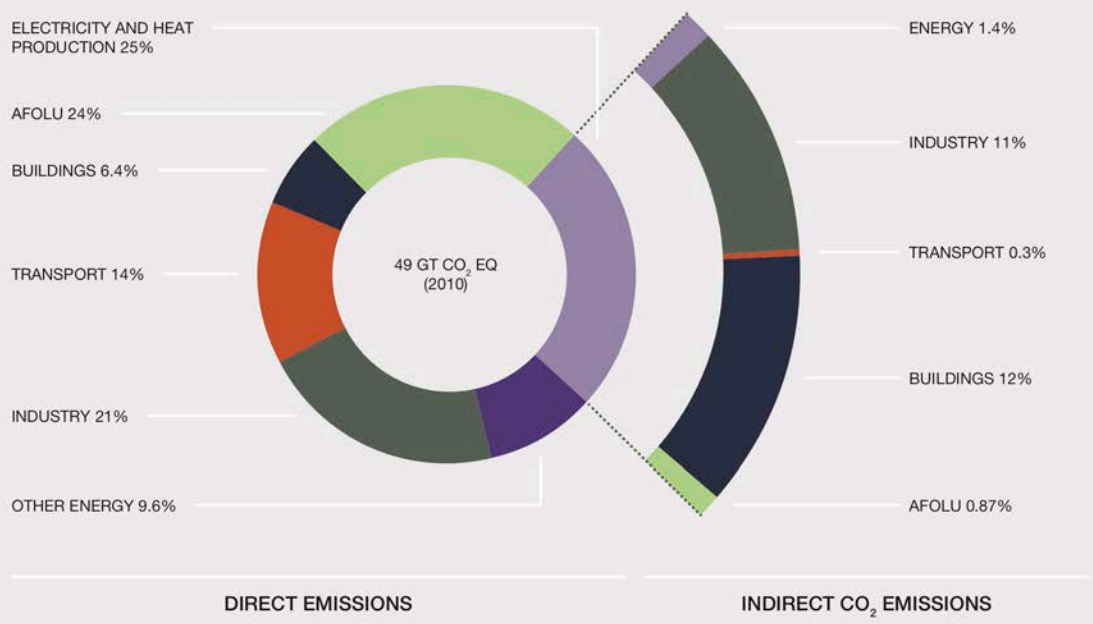
Earth’s climate, as well as the atmospheric greenhouse gases of its atmosphere, has changed throughout its history. During the pre-industrial period, the ice core shows that the greenhouse gases concentration stayed within well-defined natural limits with a maximum concentration of approximately 300 parts per million, 800 parts per billion and 300 parts per billion for carbon dioxide, methane and nitrous oxide, respectively, and a minimum concentration of approximately 180 parts per million, 350 parts per billion and 200 parts per million.

The last report of the Intergovernmental Panel on Climate Change (IPCC) (IPCC, 2014a) indicates that greenhouse gasses from anthropogenic sources have significantly increased since the pre-industrial era because of economic and population growth. This has led to atmospheric concentrations of carbon dioxide, methane and nitrous oxide that are unprecedented in at least the last 800,000 years. The IPCC reports that this significant increase in greenhouse gasses has caused a warming of 0.85ºC on average globally (land and ocean surface combined) over the period 1880 to 2012. The most recent report of the World Meteorological Organization stated that the warming has now exceeded 1^{o}C.

As shown in Figure 4.10, the economic sectors that contributes the most to greenhouse gasses are the electricity and heat production sector, agriculture, forestry and other land use, the industry sector, and the transport sector (emissions are converted into carbon dioxide-equivalents based on Global Warming Potential (100) from the IPCC Second Assessment Report) (IPCC, WGIII, 2014).

The IPCC developed the representative concentration pathways (RCPs) as a way of projecting how factors like population size, economic activity, lifestyle, energy use, land use patterns, technology and climate policy, will have an impact in the concentration of atmospheric greenhouse gasses. There are four RCPs: a stringent mitigation scenario (RCP2.6) (this scenario is based on the goal of maintaining global warming below 2ºC above pre-industrial temperatures), two intermediate scenarios (RCP4.5 and RCP6.0) and one scenario with very high greenhouse gasses emissions (RCP8.5) (IPCCC, 2014b).

Figure 4 10 Greenhouse gas emissions by economic sectors. Source: IPCC (2014).



The IPCC (Stocker et al., 2013) reported that in all of these scenarios, except RCP2.6, global surface temperature change for the end of the 21st century is likely to exceed 1.5 °C relative to 1850 to 1900. Furthermore, under two scenarios (RCP6.0 and RCP8.5) it is likely that global surface temperature change will exceed 2°C (the upper limit of the goal of the Paris Agreement), and more likely than not to exceed 2°C for RCP4.5. (IPCC, 2013).

Mean surface temperatures for 2081-2100 relative to 1986-2005 is likely to increase in the following ranges for each scenario: 0.3°C to 1.7°C (RCP2.6), 1.1°C to 2.6°C (RCP4.5), 1.4°C to 3.1°C (RCP6.0), 2.6°C to 4.8°C (RCP8.5) (IPCC, 2013).

Moreover, it is very likely that heat waves will occur more often and last longer and that extreme precipitation events, both floods and droughts, will become more intense and frequent in many regions (IPCC, 2013).

The ocean will continue to warm. In the top hundred meters, ocean warming is expected to be about 0.6°C (RCP2.6) to 2.0°C (RCP8.5), and about 0.3°C (RCP2.6) to 0.6°C (RCP8.5) at a depth of about 1,000 meters by the end of the 21st century (IPCC, 2013).

Global mean sea level will continue to rise during the 21st century, with the rate of rise very likely exceeding that observed during 1971 to 2010 due to increased ocean warming and increased loss of mass from glaciers and ice sheets. Sea level rise for 2081–2100 relative to 1986–2005 will likely be in the ranges of 0.26 to 0.55 meters for RCP2.6, 0.32 to 0.63 meters for RCP4.5, 0.33 to 0.63 meters for RCP6.0, and 0.45 to 0.82 meters for RCP8.5. For RCP8.5, the rise by the year 2100 is 0.52 to 0.98 meters, with a rate during 2081 to 2100 of 8 to 16 millimeters per year (IPCC, 2013).

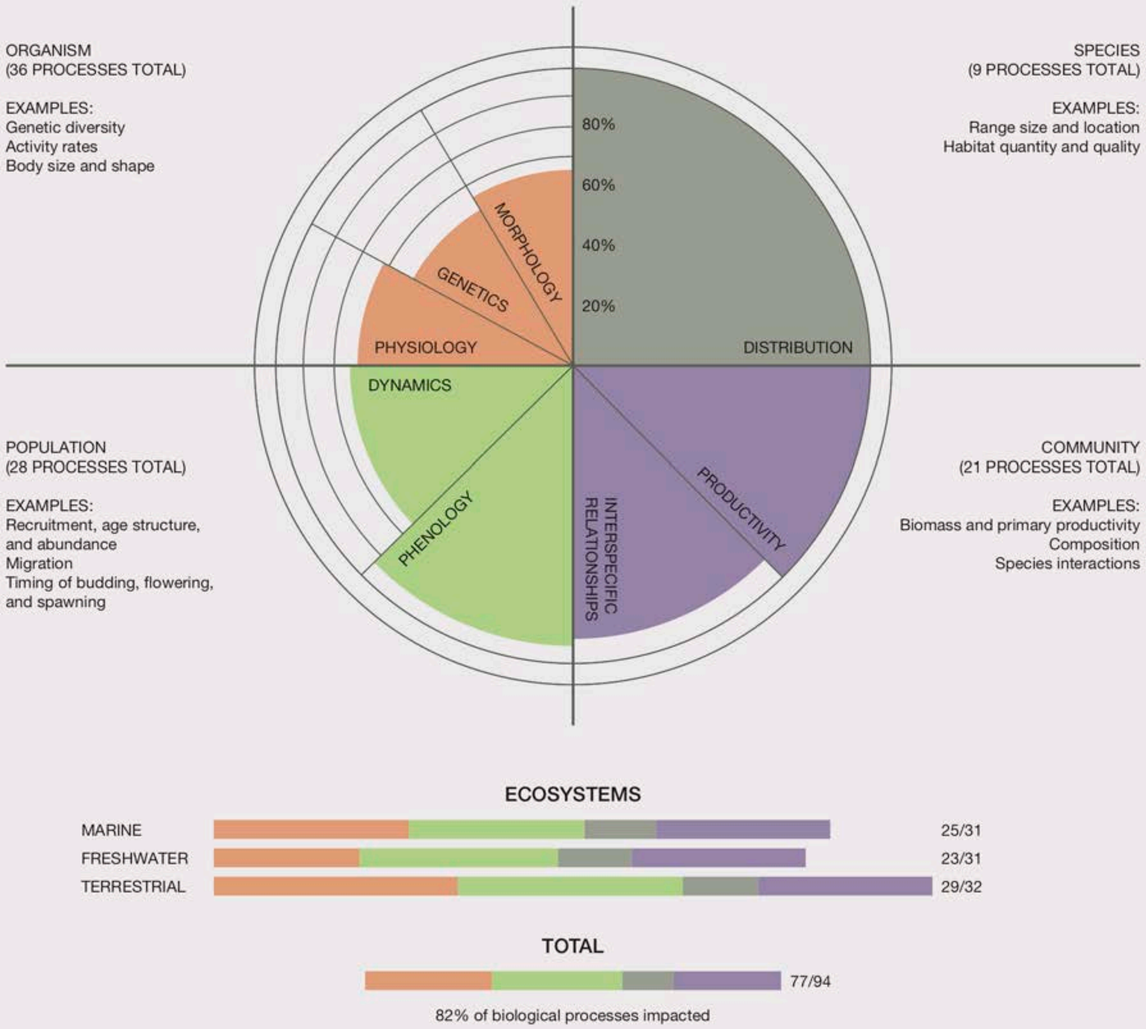
Biodiversity is impacted significantly by climate change in a wide range of ways and scales (i.e. ecosystems, species, genes). Scheffers et al. (2016) identified a set of 32 core terrestrial ecological processes and 31 each in marine and freshwater ecosystems that supports ecosystem functions and its capability in providing benefits to people. From this set of 94 processes, the authors state that 82% show evidence of impact from climate change like shifts in species ranges, changes in phenology and population dynamics, and disruptions from the gene to the ecosystem scale (Scheffers et al., 2016) (Figure 4.11).

In order to illustrate the impact of climate change on biodiversity, the following is a summary based on the findings of the last report of the IPCC on impacts, adaptation and vulnerability. In general, many terrestrial, freshwater, and marine species have shifted their geographic ranges, seasonal activities, migration patterns, abundances, and species interactions in response to climate change (IPCC, 2014a).

Certain naturally occurring factors, like the El Niño Southern Oscillation, have the potential to exacerbate the effects that climate change is already having in many parts of the Americas region. The El Niño Southern Oscillation warming and cooling phases (i.e., El Niño and La Niña, respectively) are known to predictably alter precipitation and temperature patterns both spatially and temporally throughout the region. Between December and January, El Niño generally causes wetter conditions in southwestern portions of North America (northwestern Mexico and southwestern USA), northwestern portions of South America (Colombia, Ecuador, and Peru), drier conditions in the Amazon basin, and warmer conditions in southeastern Brazil and the northeastern and northwestern portions of North America (NOAA, 2016). Between June and August, El Niño can be associated with drier and warmer conditions in Central America, wetter conditions in central Chile and the northwestern USA, and warmer conditions on the east and west coasts of central South America (NOAA, 2016). Consequently, areas experiencing drier conditions as a result of climate change, like the tropical dry forest in Central America (Fuentes-Franco et al., 2015), may experience intensified conditions during El Niño years.

Figure 4 11 **Ecological process in terrestrial, marine and freshwater ecosystems that will be impacted by climate change.**

Impacts are measured on multiple processes at different levels of biological organization within ecosystems (i.e. organism, species, population and community). Source: Scheffers *et al.* (2016).



Extreme weather events, like coastal storms, can intensify the effects that climate change-related sea-level rise is already having on many coastal areas. Specifically, coastal regions that exist in low-lying areas and are already experiencing inundation from sea-level rise are especially vulnerable to storm surge from tropical cyclones (i.e. hurricanes, typhoons), which increases flooding and land subsidence (Yang et al., 2014). Areas in the Americas region that are particularly susceptible to both sea-level rise and tropical cyclones include coastlines and island nations/territories in the Caribbean Sea, Gulf of Mexico, north Atlantic Ocean (along the southeastern coast of the USA), and northeast Pacific Ocean (along the western coast of Mexico).

Terrestrial and freshwater ecosystems

Under all the RCP scenarios, the extinction risk of a large fraction of terrestrial and freshwater species by climate change in the 21st century and beyond is increased by the interaction of other drivers of biodiversity loss like pollution, habitat modification, over exploitation, and invasive species. These ecosystems will be at risk of abrupt and irreversible regional-scale change in the composition, structure, and function under medium- to high-emissions scenarios.

Climate changes exceeding those projected under RCP2.6 in high-altitude and high-latitude ecosystems will lead to significant changes in species distributions and ecosystems function. The increase in water temperature due to global warming will lead to shifts in freshwater species distributions.

For the second half of the 21st century, all the RCP scenarios indicate that the composition of communities will change due to a change (decrease or increase) in abundance of some species, and that the seasonal activity of many species will change differentially, causing the disruption of life cycles and interactions between species. In addition, human health will be affected as a consequence of the change in the distribution (in altitude and latitude) and/or abundance of certain organisms that are important disease vectors (in fewer cases the capacity of vectors will be reduced) (IPCC, 2014b).

Climate change will reduce the populations, vigour, and viability of species with spatially restricted populations (e.g. small and insulated habitats and mountaintops). Extinctions of endemic species could be as high as 39-43% (i.e. >50,000 plant and vertebrate species) under worst case scenarios (Malcom et al., 2005)

Marine ecosystems

As in terrestrial and freshwater species, some marine species will change their distribution due to the projected warming of the planet, causing high-latitude invasions and local-extinction rates in the tropics and semi-enclosed seas (Muhling et al., 2015; Liu et al., 2015). The economic dimension of these changes is different across the world, where species richness and fisheries catch potential are projected to increase (on average) at mid and high latitudes, contrary to what would happen in tropical latitudes.

For example, the IPCC (Field et al., 2014) states that in North America there is going to be a shift in distribution of the northwest Atlantic fish species, changes in mussel beds along the west coast of the USA, and a change in migration and survival of salmon in northeast Pacific. In South America, mangrove degradation on the north coast will be impacted in a minor scale by climate change (pollution and land use are the main drivers of change). In the polar regions, climate change will significantly impact Arctic non-migratory species, the reproductive success of Arctic seabirds, populations (decrease) of southern ocean seals and seabird populations, thickness of foraminiferal shells (reduction) in southern oceans due to ocean acidification, and the density of krill (reduced) in the Scotia Sea.

Three main drivers related to climate change and emissions of carbon dioxide will have a negative impact on coastal ecosystems: 1. Sea level rise, which is related to the capacity of animals (e.g. corals) and plants (e.g. mangroves) to keep up with the vertical rise of the sea; 2. Ocean temperature, which has a direct impact on species adjusted to specific and sometimes narrow temperature ranges (e.g. coral bleaching). As a response to warmer temperatures, many marine species change their distributions towards the poles; 3. Ocean acidity, caused by the absorption of carbon dioxide that produces carbonic acid. An increase of acidity in seawater diminishes the ability of “calcifiers” (e.g. shellfish, corals) to produce carbonate to make their shells and skeletons.

The physical, chemical, and biological properties of the ocean will be altered by climate change, causing a change in the physiological performance of marine biodiversity.

Shifts in populations, geographic distribution, migration patterns, and phenology of species caused by climate change, have been and will be paralleled by a reduction in their maximum body size. Furthermore, this has caused and will continue causing a change in the interaction between species (e.g. competition and predator-prey dynamics).

Regional changes in the temperature of the atmosphere and the ocean will be accompanied by changes in glacial extent, rainfall, river discharge, wind, ocean currents, and sea level, among many other environmental parameters. There are large fluctuations in ocean conditions in each ocean basin, like the El Niño Southern Oscillation, the North Atlantic Oscillation, and the Atlantic Multidecadal Oscillation, each leading to major changes that have impacts on the coastal zone. There are, on the other hand, very large differences in freshwater supply in different coastal locations, and processes in the watershed, including the balance of different human activities, are different in all watersheds. All of these factors work together in different ways to affect any one coastal habitat.

North America

Climate in the Arctic is harsh, characterized by cold winters and cool summers. Consequently, plant growth is restricted to a relatively short growing season on the order of three months or less during the boreal summer. The tundra biome is home to approximately 1,800 species of vascular plants and has less species diversity than more temperate biomes (Callaghan et al., 2005) (see Chapter 3 for more details). Alpine tundra can also occur at high elevations in mountain ranges of North America.

Global temperature increases during the twentieth century have been amplified in the Arctic, with mean annual temperature increases approximately twice that of the global increase. For example, over the past 60 years, Alaska has warmed more than twice as rapidly as the rest of the USA, with state-wide average annual air temperature increasing by 1.7 °C and average winter temperature by 3.4 ºC, with substantial year-to-year and regional variability (Chapin et al., 2014). The overall warming has involved more extremely hot days and fewer extremely cold days.

There is increasing evidence that physical and ecological changes are already occurring throughout the tundra biome (Hinzman et al., 2005; McGuire et al., 2006), and includes increases in photosynthetic activity (Bunn & Goetz, 2006) and an expansion of shrub tundra at the expense of graminoid tundra (Myers-smith et al., 2011).

Average annual temperatures in the northern tundra region of Alaska are projected to rise by an additional 2.5 ºC to 5 ºC by the end of this century depending on fossil fuel emissions (Chapin et al., 2014). Annual precipitation is projected to increase about 15% to 30% by late this century if global emissions continue to increase (Chapin et al., 2014). However, increases in evaporation due to higher air temperatures and longer growing seasons are expected to reduce water availability.

The changes in climate are projected to increase the area occupied by shrub tundra in northern Alaska by 2% to 21% by the end of this century, largely at the expense of graminoid tundra, which is projected to decrease by 8% to 24% (Rupp et al., 2016). Treeline is projected to move slightly northward in some climate scenarios (see Chapter 3 for more details). Climate change is also expected to have significant consequences for the distribution and diversity of Alpine tundra ecosystems in mountain ranges of North America, as tundra ecosystems may shift to higher elevations and lose biodiversity (Lesica, 2014).

Notably, the acceleration in ice sheet loss over the last 18 years was 21.9 ± 1 Gt/yr^2 for Greenland (Rignot et al., 2011). In July 2012, over 97% of the Greenland ice sheet experienced surface melt, the first widespread melt during the era of satellite remote sensing. Since Arctic temperatures are expected to rise with climate change, the authors' results suggest that widespread melt events on the Greenland ice sheet may begin to occur almost annually by the end of century (Keegana et al., 2014). Lenton (2011) included the irreversible melt of the Greenland ice sheet as one of the eight candidates of human-induced climate change tipping points. Biodiversity and ecosystem services of Greenland are highly vulnerable to anthropogenic climate change (Larsen et al., 2014).

Boreal forests and temperate forests: warming in the boreal forest area of Alaska has occurred throughout the 20th century, with mean annual temperatures increasing between 0.5 and 3.0 ºC in regions south of 60 ºN (Price et al., 2013). Since 1900, annual precipitation amounts appear to have increased by 10% to 20% throughout much of the boreal zone of Canada, although drought conditions have existed in western Canada since 1995 (Price et al., 2013). In the temperate zone of North America, warming has also been substantial (~0.9 ºC since 1895, Melillo et al., 2014). In recent decades, moisture availability has decreased in the southeast and west, while the northeastern USA has experienced more extreme precipitation events (Melillo et al., 2014). These changes in climate in recent decades have generally increased tree mortality of both boreal and temperate forests through fire, insect infestations, drought, and disease outbreaks (Price et al., 2013; Chapin et al., 2014; Joyce et al., 2014).

Annual mean temperatures across the Canadian and Alaska boreal zones are projected to be 4 to 5 ºC warmer by 2100 (Price et al., 2013; Chapin et al., 2014). Although annual precipitation is projected to increase in Canada and Alaska, increases in evaporation due to higher air temperatures and longer growing seasons are expected to reduce water availability to these forests. In the temperate zone, another 1 to 2 ºC warming is expected by 2100, with continued reduced water availability in the southeast and western USA (Melillo et al., 2014). Although climate envelope models for individual species suggest that these changes could potentially result in substantial shifts in species ranges in response to climate change, they generally do not account for limiting factors such as soil suitability, geographic barriers, and seed dispersal distances, which all limit the rate at which new areas can be colonized (Price et al., 2013). The application of models that do consider these limiting factors indicate that northward migration of boreal forest into tundra regions will be very limited during the remainder of this century (Rupp et al., 2016). However, the projected climate changes for North America are expected to increase the vulnerability of boreal and temperate forest to increased mortality through fire, insect infestations, drought, and disease outbreaks, particularly in areas where water availability is already a concern (Price et al., 2013; Chapin et al., 2014; Joyce et al., 2014). For example, the analyses of Rupp et al. (2016) estimate that changes in the fire regime will decrease late successional boreal conifer forest by 8% to 44% by the end of this century, with a concomitant increase in early successional deciduous forest. In lowland forest areas of the boreal zone underlain by ice-rich permafrost, forest mortality could increase because of subsidence and inundation associated with permafrost thaw (Price et al., 2013). However, in both boreal and temperate forests with well-drained soils and adequate water availability, it is expected that forest productivity may increase (Price et al., 2013; Joyce et al., 2014).

Increasing temperatures and changes in the amount and timing of precipitation are expected to affect the temperate grasslands of North America. However, despite potential increases in aridity, particularly during summer, the fractional cover of green foliage may increase under future climate scenarios (Hufkens et al., 2016). This increase is likely to occur from earlier spring green-up and later autumn senescence, which may more than compensate for any reduction of fractional cover during hot, dry summers (Hufkens et al., 2016).

Many of the dryland regions of North America are experiencing changes in climate. The Great Basin, Colorado Plateau, Mojave in the USA and Sonoran Desert in northwestern Mexico and the southwestern USA have experienced a warming trend, particularly during winter and spring, and the freeze-free season has lengthened (Weiss & Overpeck, 2005; Cook & Seager, 2013). These temperature changes have the potential to shift vegetation types northward and eastward and upward in elevation (Weiss & Overpeck, 2005), having implications for the adjacent deserts (Notaro et al., 2012).

Wetlands in the Prairie Pothole region (freshwater marshes, wet meadows, etc.) are experiencing increased temperatures and variability in precipitation, which may have implications for waterfowl and important ecosystem services. Projected changes in temperature and precipitation of more than 1.5-2.0 °C may diminish wetland function across the majority of the Prairie Pothole region (Johnson & Poiani, 2016).

Northern portions of the Everglades in South Florida are dominated by peatlands that depend on adequate amounts of precipitation to balance the constant loss of water through evapotranspiration, but increased periods of drought have the potential to cause large shifts in plant and animal communities (Nungesser et al., 2015). In southern portions of the Everglades, plant communities are threatened by increased salinity from sea level rise, which can create physiological drought and a shift from freshwater to saltwater-tolerant species (Nungesser et al., 2015). In the Florida region, models and field data indicate that mangrove forests will continue to expand their latitudinal range as temperature and atmospheric carbon dioxide concentrations increase (Alongi, 2015).

Average annual temperatures have increased by as much as 0.25°C per decade since the middle of the twentieth century in some parts of the Great Lakes region of North America (Hayhoe et al., 2010). Those temperature changes have advanced the timing of spring, lengthened the growing season (Robeson, 2002), and produced low lake levels (Notaro et al., 2015a).

The frequency of heavy rainfall events has nearly doubled since the 1930's (Angel & Huff, 1997; Kunkel et al., 1999; Villarini et al., 2011) and is associated with hydrologic flooding in some areas of the midwest (Peterson et al., 2013). Increased lake surface temperatures, frequent and intense cyclones, and reduced ice cover have been associated with more occurrences of lake-effect snow (Burnett et al., 2003; Kunkel et al., 2009), which can affect hydrologic systems and species that are sensitive to changing moisture regimes (Davis et al., 2000; Burnett et al., 2003). Warming lake temperatures have been shown to generate low oxygen conditions in deeper portions of the lakes and extreme precipitation and drought events may play a role in harmful algae growth (Zhou et al., 2015), both affecting fish growth, reproduction, and survival (Scavia et al., 2014). Additionally, warming lakes have been shown to alter the extent and duration of temperature preferences for some commercially important fish species, potentially intensifying competition and food-web interactions (Cline et al., 2013).

Ice cover in the Great Lakes is projected to continue declining and will eventually be restricted to the northern lake shores in mid- to late winter (Notaro et al., 2015b). Enhanced evaporation from lack of ice cover will increase lake-effect precipitation, but it will consist primarily of rain due to increasing temperatures (Notaro et al., 2015b). However, because both precipitation and evaporation over lakes is expected to increase, the influence on lake levels is still unclear (Angel & Kunkel, 2010; Notaro et al., 2015a).

The pelagic ocean is presenting changes in major wind patterns, ocean currents, temperature, and pH (e.g. Bates et al., 2014; Muller-Karger et al., 2015). For example, it is expected that the north Atlantic Ocean will continue the warming trend that has been observed there over the past decade (Liu et al., 2015, 2016). These changes are expected to have an impact on suitable habitat of a number of valuable fish and affect fisheries that depend on them (Kerr et al., 2009; Hare et al. 2010, Lenoir et al., 2011; Muhling et al., 2015, 2017). Warming off the Alaska coast since the late 1970s triggered a decline in forage species (e.g. shrimp and capelin) and an increase in high-trophic level groundfish (Anderson & Piatt, 1999). This community reorganization negatively affected seabirds, marine mammals, and other species that depend on forage species (Anderson & Piatt, 1999). A warm-water anomaly (i.e. "the blob") was detected off the Alaska coast during the winter of 2013-2014, with near-surface temperatures 2.5°C greater than normal that eventually stretched south to Baha, California (Bond et al., 2015; Cavole et al., 2016). The cause of the anomaly is believed to be the result of reduced heat exchange between the ocean and the atmosphere and weak horizontal advection in the upper ocean, which may have been triggered by a much higher than normal sea level pressure (Bond et al., 2015). The anomaly negatively affected commercially-important fisheries, including tuna, and was responsible for marine mammal and seabird strandings (Cavole et al., 2016).

Mesoamerica

Precipitation is projected to decline during the wet season throughout the region and mountainous areas in Costa Rica and Panama, which generally receive a large amount of orographic moisture, will see a decline in precipitation (Karmalkar et al., 2011). Differential warming of the Pacific and Atlantic sea surface temperatures, which causes a stronger Caribbean low level jet, will lead to drier conditions in Mexico and Central America (as much as 50% drier) during summer (Fuentes-Franco et al., 2015) and has the potential to lead to water stress in many regions. Additionally, severe and extended dry seasons are likely to lead to forest species turnover and loss of many tree species (Condit, 1998). However, Prieto-Torres et al. (2015) found that while tropical dry forests are projected to decline in many areas of Mexico, they may increase in other areas by moving upward in elevation.

Changes in temperature and precipitation have the potential to affect the climate-sensitive cloud forests of Mesoamerica by causing biodiversity loss and shifts from the unique ecosystems to lower-altitude vegetation types (Foster, 2001). Additionally, climate changes may result in changes in cloud formations, which are already being observed in certain parts of Costa Rica (Foster, 2001). Although sea evaporation is likely to increase with increasing sea surface temperatures, pumping more water into the atmosphere, cloud formation is expected to increase in height, which will alter the relative humidity and amount of sunlight the forests are exposed to (Foster, 2001). The total area of cloud forests in Mexico is expected to decline by as much as 70% by 2080 (Ponce-Reyes et al., 2013). However, models suggest that minimizing land-use change and developing protected areas in remaining cloud forests may promote dispersal and allow some species to persist despite changes to climate (Ponce-Reyes et al., 2013). In addition, protected areas can have other benefits, such as the ability to capture and reduce carbon dioxide emissions into the atmosphere (Uribe, 2015).

The Mesoamerican tropical dry forests are experiencing increased warming (Aguilar et al., 2005, Karmalkar et al., 2011). Between 1961 and 2003, the percentage of warm minimum and maximum temperatures have increased by 1.7% and 2.5% per decade, respectively, whereas the percentage of cool minimum and maximum temperatures have decreased by 2.4% and 2.2% per decade, respectively (Aguilar et al., 2005). Most of the precipitation in the tropical dry forests occurs during the summer (Fuentes-Franco et al., 2015) and is likely an important factor in the distribution of tropical tree species richness (Somers et al., 2015). Although no trend in the amount of precipitation has been observed, the intensity of rainfall events has increased over the last 40 years (Aguilar et al., 2005).

Karmalkar et al. (2011) projected that warming in the region will vary both spatially and temporally, with higher temperatures in the Yucatan Peninsula and during the wet season. Increased temperatures in the tropical dry forest has implications for carbon sequestration, as carbon uptake is likely to decline substantially under warming conditions (Dai et al., 2015). Additionally, because understory microsite variability is low in some portions of the tropical dry forests, future warming could have serious implications for neotropical birds (Pollock et al., 2015). A temperature increase >3°C has the potential to cause a 15% decline in potential species richness (Golicher et al., 2012) (see Chapter 3 for more details).

Most wetlands in Mexico are found along the Gulf of Mexico or Pacific Ocean (Mitsch & Hernandez, 2013). Similarly, mangrove swamps are common on both coastlines in Central America (Mitsch & Hernandez, 2013). Consequently, sea level rise is by far one of the largest concerns with regards to climate change impacts on wetland resources in those regions (Mitsch & Hernandez, 2013). The effects of sea level rise on mangrove ecosystems, for example, could have implications for fish, mollusks, and aquatic mammals (Botero, 2015). However, feedbacks between plant growth and geomorphology may allow for wetlands to maintain stability and resist the negative impacts of sea level rise. This resiliency likely depends on human interference, such as groundwater withdrawal or artificial drainage of wetland soils, which can lead to more rapid subsidence (Kirwan & Megonigal, 2013). Additionally, the construction of dams and reservoirs may prevent sediments needed for wetland building from reaching coastal areas, which can minimize the likelihood for wetland sustainability under sea level rise (Kirwan & Megonigal, 2013).

Caribbean

Most insular ecosystems in the Caribbean Sea have experienced a warming trend in recent decades, with increases in both daily minimum and maximum temperatures (Karmalkar et al., 2013). However, those trends vary by region as Puerto Rico has experienced an increase in daily minimum temperatures, but a decrease in daily maximum temperatures (Van Beusekom et al., 2015).

Ecosystems found in Caribbean regions may be particularly vulnerable to rising sea levels; Bellard et al. (2014) projected that 63 out of 723 Caribbean islands would be completely submerged with 1 m of sea-level rise and 356 islands submerged with 6 meters of sea-level rise, which may have implications for hundreds of endemic species inhabiting the islands. Additionally, tropical cyclones are expected to increase in intensity (as well as frequency of intense storms) as a result of climate change (Michener et al., 1997; Reyer et al., 2015). Some regions of the Caribbean may receive a large proportion of their annual rainfall from hurricanes (Scatena & Larsen, 1991), which may be important given droughts increased between 1950 and 2010 (Dai, 2012). The frequency of droughts is also expected to increase in the future (Reyer et al., 2015). Karmalkar et al. (2013) estimated that precipitation is likely to decline by 5.7% to 24.6% (depending on the model) between the years of 2080 and 2089 compared with 1970 and 1989. Reduced preciptitation, along with warmer temperatures, have the potential increase evapotranspiration and drought risk (Reyer et al., 2015).

The region’s forests and terrestrial biodiversity are also threatened by climate change (see Chapter 3 for details). While hurricanes are part of the Caribbean’s “normal” environment and ecosystems have adapted to them, the repeated and compounding impacts of frequent extreme weather events has been shown to reduce their ability for recovery. The flash floods and mudslides that caused the many fatalities during the devastating 2008 hurricane season in Haiti, would probably not have been so severe had the mountains not been deforested. Protecting forests and improving their resilience will be an important adaptation strategy both for the conservation of biodiversity and for the future wellbeing of Caribbean communities (Day, 2009).

Warming of coastal areas has had marked impacts on the population, diversity, and health of coral reef resources in the Caribbean Sea and Gulf of Mexico (Eakin et al., 2010; Vega-Rodriguez et al., 2015; van Hooidonk et al., 2015). Increased water temperatures have the potential to affect fisheries in Caribbean countries. Cheung et al. (2010) estimated that catch potential off Caribbean coasts may decrease as much as 5% to 50% between 2050 (2°C of warming) and 2100 (4°C of warming).

The global net value of the coral reefs of the Caribbean Sea services related with fishery, coastal protection, tourism, and biodiversity, were estimated $29,800 million per year. Currently two thirds of the Caribbean coral reefs are impacted detrimentally by human activities, including climate change, (GEO 4, UNEP, 2007).

Mass coral bleaching events have also become more frequent and more severe in recent years as a result of increasing sea surface temperatures and aragonite saturation, in particular the widespread and catastrophic bleaching event of 2005 in the Caribbean. This is presenting a new challenge to islands dependent on reefs for fisheries, dive tourism and coastal protection (Day, 2009). By 2050, with 1.5°C to 2°C, there is 20-40% to 60-80% probability, respectively, that coral reefs in the Caribbean and western Atlantic will undergo yearly bleaching events (Meissner et al., 2012). Nearly all coral reefs are expected to undergo severe bleaching by 2100, with exception to areas with upwellings (Meissner et al., 2012).

The IPCC (2014) considers the small island states, like those of the Caribbean, to be among the most vulnerable to the projected impacts of climate change, like rising sea levels, intensifying storms, mass coral bleaching events, ocean acidification, and potential water and food shortages.

South America

Although the Amazon basin has experienced periodic warming and cooling since the 1900s, which may be associated with the Pacific Decadal Oscillation (Malhi & Wright, 2004; Gloor et al., 2015), annual mean temperature has steadily increased since the 1970s (Victoria et al., 1998, Malhi & Wright, 2004; Vincent et al., 2005) and is more intense during the dry season than the wet season (Gloor et al., 2015). Trends in long-term precipitation patterns and their link to climate change (as opposed to Pacific Decadal Oscillation and El Niño Southern Oscillation) are less clear (Marengo, 2004; Satyamurty et al., 2010). However, Gloor et al. (2015) showed that although annual net rainfall has increased in the area, the amount of rainfall during the dry season has decreased since the 1970s. Those trends are concerning given that droughts in the tropical forests have been associated with reduced vegetation growth and browning (de Moura et al., 2015), slow canopy recovery times (Saatchi et al., 2013), reduced above ground live biomass (Saatchi et al., 2013), and accelerated tree mortality over large areas (Phillips et al., 2009). During the wet season, the frequency of heavy rainfall events and severity of Amazon flood pulses has increased (Donat et al., 2013; Gloor et al., 2015), potentially affecting the ecology of floodplain and swamp forests in the Amazon basin.

Climate projections suggest that both temperature and precipitation trends are likely to continue, with a substantial lengthening of the dry season by the end of the twenty-first century (Boisier et al., 2015). Those conditions have the potential to prevent the tropical forest distribution from moving upslope (staying restricted to wet areas) and persisting along ecotones, and could eventually cause it to convert to savannah-type vegetation in eastern portions of the basin (Olivares et al., 2015). Additionally, species richness and plant productivity are likely to decline, altering the Amazon basin from a carbon sink to a source (Olivares et al., 2015). Finally, the severity of wet-season flood pulses is projected to increase and may have implications for movement and reproduction of many Amazon River-associated species (Zulkafli et al., 2016).

There is no climatic assessment devoted exclusively to the Amazonian wetlands. However, the IPCC Regional Assessment for Central and South America (Magrin et al., 2014) covers the entire distribution of this environment. Based exclusively on this assessment in the northern part of South America, some inferences can be drawn in regard of these wetlands. The trends are:

- Temperatures: In general terms, with the exception of interior Venezuela, 30% to 50% increase in temperature is expected in northern South America, representing +5°C to +7°C. And for the period of 2071 to 2100 another increase from +4°C to +5°C is expected (Marengo et al., 2012). This problem is exacerbated in urban environments, even in small island developing states (Mendez-Lazaro et al., 2017).
- Precipitation: In general, an increase from 30% to 50% in precipitation is expected in northern South America. However, while a decrease of 20% to 30% in rainfall in central and eastern Amazonia, is expected, an increase from 10% to 30% in rainfall in western Amazonia is expected (Giorgi & Diffenbaugh, 2008; Mendes & Marengo, 2010; Sorensson et al., 2010; Marengo et al., 2012). This increase in rainfall for western Amazonia will be observed both in summer and winter. This, in turn, will deeply affect flooding patterns in wetlands in northern and western Amazonia. Effects of precipitation on current flows, rivers discharge and potential flooding was observed for most of the large rivers (Dai et al., 2009; Dai et al., 2004)
- Sea level: In coastal areas an increase in sea level is expected, with increase in flood probabilities (>40%). Impacts of flooding can be costly and coastal communities should evaluate possible solutions to cope with this problem (Marengo et al., 2017). Extreme events: Longer dry periods, or consecutive dry days, are expected for the region, with an increase of up to 8% (or 5 more dry days). Heavier precipitation in northern and western Amazonia (from 1 to 10mm) is also expected.

All impact analysis available indicates that these extreme events and the trends of climate change in Amazonian wetlands and rivers will be very strong (Marengo & Espinoza, 2016). Extreme events will be more frequent and more intense, and floods and droughts will impact both natural and human systems in the region. Although with a large range of uncertainty, wetlands in the northern and western Amazonia may experience more frequent floods, while wetlands in eastern Amazonia might be under more intense and severe droughts. These effects might cause great changes on the biota of all wetlands affected. Intense floods can bring losses in crops (inundation of small farms and gardens), in local and regional fisheries, and even in human lives. Intense droughts are associated with fire incidence, and additional aerosol emissions, public health problems, and other losses in agriculture and fisheries (Marengo & Espinoza, 2016).

Most areas in the Andes Mountains have experienced a warming trend (Vuille et al., 2015), particularly during winter (Barros et al., 2015). Magrin et al. (2014; and references therein) showed that temperatures have increased by 0.1°C to 0.6°C per decade across different regions of the Andes since the 1950s and 1960s. The warming conditions have caused many of the Andean glaciers to retreat, creating a loss of important water reserves (Barros et al., 2015). Additionally, snow is melting earlier in the spring and has affected the timing of maximum stream flows, which are peaking as much as a month earlier in recent years than when compared to the early twentieth century (Barros et al., 2015). Reduced river flows in Argentina have suggested a decrease in precipitation (Barros et al., 2015), but the precipitation trends are less clear in other regions of the Andes Mountains (Marengo et al., 2009). Vuille et al. (2003) found that precipitation was greater north of approximately 11°S, whereas stations found south of that mark showed decreasing precipitation between 1959 and 1994.

Projected temperatures suggest increases of 2.0-3.5°C by the end of the 21st century, which has the potential to cause glaciers to retreat substantially or disappear altogether (Barros et al., 2015). Precipitation is most likely going to increase between the latitudes of 5°N and 20°S, particularly in northern Peru where precipitation could increase as much as 70% (Marengo et al., 2011). However, precipitation is most likely going to decrease (as much as 10%) in the subtropical Andes south to Patagonia and on the altiplano (Marengo et al., 2011). Additionally, Andes snowfall will be less common in the mountains of Argentina and melt earlier in the spring, affecting the amount of water available for summer irrigation (Barros et al., 2015). Important tropical Andes ecosystems, like páramos, punas, and evergreen montane forests, are projected to undergo a large amount of species turnover or loss of species richness (Ramirez-Villegas et al., 2014). The páramo grasslands, glaciers, and cryoturbated areas, which are found at the highest elevations, may be at greatest risk (Tovar et al., 2013). Species found in the cloud forests of the Andes may be at risk of extinction due to observed upward shifts in ecotones, which could serve as barriers to species migration (Lutz et al., 2013) (see Chapter 3 for more details).

The Brazilian Cerrado, a large area of tropical dry forest, savanna, and grasslands found on the Brazilian Central Plateau, has been trending warmer, with an annual maximum temperature increase of 0.79°C between 1980 and 2004 (Santos, 2014). Additionally, the number of days with temperatures >25°C increased at a rate of 4.4 days per year during that same time period (Santos, 2014). Precipitation trends are less clear, with the exception of the number of days with heavy precipitation (>10mm), which showed a decrease of 0.43 days per year between 1980 and 2004 (Santos 2014). Projected temperature increases may increase as much as 2.5°C to 5.5°C over tropical and subtropical latitudes and precipitation is expected to decrease during most seasons (with exception to winter) by the end of the 21st century (Cabré et al., 2016). This warming trend along with reduced precipitation (Marengo et al., 2009) could have implications for fire activity. Fire is an important factor in the grassland regions of the Cerrado, and has increased in frequency since European settlement (Pivello, 2011). Although fire is often anthropogenic in nature, it can occur naturally through lightning strikes and is particularly destructive in areas where fire is actively suppressed, having important implications for biodiversity (Pivello, 2011). For example, small mammal communities, which play important roles in a variety of ecosystem processes (e.g. plant composition, soil structure; Sieg, 1987), have shown to be sensitive to severe fires, particularly in the savanna woodland regions of the Cerrado (“Cerradão”; Mendonca et al., 2015). Although sustainable use of fire is appropriate in the Cerrado, careful management is needed to avoid land degradation and loss of biological diversity and ecosystem processes (Pivello, 2011).

Many tropical grasslands have been targeted for reforestation to help offset carbon dioxide emissions. However, not all grassland regions are the result of deforestation and converting them to plantations has the potential to cause substantial losses in biodiversity (Bond, 2016).

Temperature are expected to increase in the Río de la Plata grasslands, particularly during spring (Cabré et al., 2016). Although precipitation in many areas of the region has been linked with El Niño Southern Oscillation (Ropelewski & Halpert, 1987), trends suggest that rainfall has increased in Uruguay, Paraguay, northern Argentina, and southern Brazil between 1960 and 2000 (Haylock et al., 2006). However, Haylock et al. (2006) found that those precipitation trends closely align with a trend towards a more negative southern oscillation index, suggesting that more frequent El Niño Southern Oscillation-like events are responsible for recent changes in precipitation. Rainfall is expected to increase in southern Brazil, particularly in summer and fall, and will decrease during winter and spring (Cabré et al., 2016). Precipitation is associated with net primary productivity in some areas of the Río de la Plata region, particularly in native forests and afforested areas, but other land use activities can interact with climate factors and cause carbon storage to decline (Texeira et al., 2015). An increase in precipitation may cause flooding, erosion, and increased nutrient runoff, which can affect biological communities in pampean rivers and streams by increasing the number of species that better tolerate turbid and enriched environments (Capitulo et al., 2010).

Climate change is likely to have a substantial impact on mangrove ecosystems (Ellison, 2015), through processes including sea level rise, changing ocean currents, increased storminess, increased temperature, changes in precipitation, and increased carbon dioxide. Exposure to disturbances induces dynamism on annual and decadal scales that is reflected in changes in the populations, biomass, and spatial distribution of the mangrove ecosystem (Schaeffer-Novelli et al., 2016). Sea level rise is likely to influence mangroves in all regions, although local impacts are likely to be more varied. Mangroves are likely to be less affected by sea level rise in areas with high sediment availability, uplifting or stable coasts, high productivity, and large tidal ranges (Ward et al., 2016), as well as along wet tropical coasts and/or in areas adjacent to significant river input (Alongi, 2008), like the Amazon estuary and Parnaiba delta.

These factors combined with increased temperatures at the latitudinal extremes of mangrove distribution, a predicted increase in the strength and frequency of El Niño events that lead to below normal rainfall and a decrease in extreme precipitation events in most of tropical South America, and a resultant decrease in the cooling and drying influence of the Humboldt Current in western South America, could provide an increase in the distribution of mangroves within South America. However, in semiarid regions of South America, where mangroves typically occur in estuaries, and irrigation and damming are more prevalent, mangroves are likely to suffer from increases in salt-stress and resultant decreases in productivity combined with decreases in sediment input (Ward et al., 2016).

Climate change mitigation and adaptation strategies

Because of the substantial increase of atmospheric greenhouse gases in recent decades, it is important to identify actions that may reduce emissions through mitigation efforts. Many mitigation policies have already been implemented in the Americas region. For example, although no national climate legislation exists, a variety of policies and measures that lower emissions have been implemented at multiple governmental levels in the USA (U.S. National Climate Assessment, 2014). Additionally, developing countries, like Brazil, are also making strides with regards to mitigation, pledging to reduce greenhouse gas emissions by as much as 40% below 2005 levels by 2030 (Brazil Intended Nationally Determined Contribution, 2015). Some communities are taking the important step of talking about possible impacts of sea level rise, for example (Marengo et al., 2017). However, because climate change is a global issue, it is important that countries work collaboratively to develop emission reduction strategies as opposed to each country approaching the problem independently (IPCC, 2014a).

Mitigation can also refer to enhancing the capacity for carbon storage in regions that may be able to remove greenhouse gases from the atmosphere (IPCC, 2014a). Both oceans and vegetated regions have the potential to serve as carbon dioxide sinks, and improving our understanding of the various physical and biological processes that can increase carbon uptake will assist with developing better estimates of potential carbon offsets. For example, it is well known that vegetated coastal regions (e.g. salt marshes, mangroves) can be important regions for carbon sequestration, but recent work has indicated that microalgae may also sequester substantial amounts of carbon and is able to deliver it to sediments and the deep sea for long-term storage (Krause-Jensen & Duarte, 2016). Similarly, calculating more accurate carbon offsets in forests requires consideration of both the ability to regulate greenhouse gases, as well as regulation of water and energy (Anderson-Taixeira et al., 2012).

Although mitigation is critical for reducing greenhouse gas emissions, the IPCC has warned that projected climate change is expected to affect human and natural systems despite the scale of mitigation policies that are adopted in the next few years (IPCC, 2007). Therefore, developing and implementing effective adaptation strategies will be needed to minimize those potential climate change impacts (IPCC, 2007). Adaptation planning is occurring in both the public and private sectors throughout many regions of the Americas. For example, many municipalities in North America are considering incremental changes to their planning efforts as a result of climate change and some regions in Central and South America are considering ecosystem-based approaches, such as developing protected areas (IPCC, 2014a). Despite increased recognition of the importance of adaptation planning in response to climate change, few measures have actually been implemented on the ground (IPCC, 2014a). Barriers to implementation include limited funding, policy and legal impediments, and difficulty in anticipating climate related changes at local scales (U.S. National Climate Assessment, 2014).

The majority of adaptation planning is focused on risk and water management and the importance of ecosystem-based adaptation is only recently being recognized (IPCC, 2014a). Vignola et al. (2009) found that developing countries, in particular, depend heavily on ecosystem services and it is critical that they be mainstreamed into national and international adaptation policies. Additionally, those authors suggested that adaptation needs to be more closely linked with mitigation to ensure certain mitigation policies are less likely to have negative impacts on the well-being of certain communities (Vignola et al., 2009). Ongoing monitoring is therefore crucial to develop a better understanding of, and adaptation to future changes. This will also allow for more effective incorporation of ecosystems into spatial planning, including disaster risk reduction strategies (UNEP, 2014). Indigenous and local knowledge also contribute to climate change mitigation and adaptation as presented in Box 4.12.

Box 4.12. Indigenous and traditional knowledge on climate change

The Millennium Ecosystems Assessment (2005) considers the traditional knowledge, or practitioners' knowledge held by local resource managers, can be of equal or greater value for ecosystem management, not only the formal scientific information.

For North America, the government agencies incorporated the indigenous communities into established initiatives to develop no-regrets and co-benefits climate change adaptation strategies. Rural and indigenous community members possess valuable local and experiential knowledge regarding NCP (Romero-Lankao, 2014).

For the Caribbean islands, the preservation of the traditional knowledge of biodiversity is crucial to the sustainable use of NCP. The loss of such traditional knowledge, for example that related to medicine plants and agriculture, has had a direct negative effect on biodiversity and on the degradation of ecosystems (Suárez et al., 2008). There is continuing strong support for the incorporation of indigenous knowledge into adaptation planning on small islands (Nurse et al., 2014).

There is a growing acknowledgement that indigenous and traditional knowledge has the potential to bring solutions to face the rapidly changing climate and that land ownership and authority of indigenous groups can help better manage many natural areas and reduce deforestation of the Central and South American region. Linking indigenous knowledge with scientific knowledge is crucial for the adaptation process, currently there is limited scientific literature discussing that subject (Magrin et al., 2014). The concept of "mother earth" (madre tierra in Spanish) as a living system has emerged in different forms in recent years, as a key sacred entity on the view of indigenous nations and as a system that may be affected by and also resilient to climate change.

Climate change is a central element of the Aichi targets of the CBD Strategic Plan for 2011-2020 (Box 4.13).

Box 4.13. Climate change and the Aichi targets of the CBD Strategic Plan for 2011-2020

The CBD recognizes the urgency of addressing climate change in order to halt the rate of biodiversity loss, and this is reflected in its Strategic Plan for 2011-2020. Because of the broad impact of climate change, this driver is covered and/or impacted indirectly by many of the Aichi targets of the Plan, in targets like number 5 on the half of natural habitats rate loss, number 11 on terrestrial and coastal and marine areas protection and number 14 on the restoration and protection of ecosystem services, to mention a few. The achievement of these targets will help to mitigate and adapt to climate change, both from an anthropocentric and biodiversity perspective.

Nevertheless, targets 10 and 15 refers directly to climate change.

Aichi target 10: By 2015, the multiple anthropogenic pressures on coral reefs, and other vulnerable ecosystems impacted by climate change or ocean acidification are minimized, so as to maintain their integrity and functioning. The aim of the target is to reduce the impact of other drivers (like the ones covered in this chapter) on vulnerable ecosystems in order to make them more resilient to the unavoidable effects of climate change. This target has a link with target 12 on the conservation of endangered species and target 15 on ecosystems resilience and carbon stocks[1].

Aichi target 15: By 2020, ecosystem resilience and the contribution of biodiversity to carbon stocks have been enhanced, through conservation and restoration, including restoration of at least 15 per cent of degraded ecosystems, thereby contributing to climate change mitigation and adaptation and to combating desertification. Carbon sequestration refers in this target to the carbon taken and stored in biomass and soils of ecosystems like tropical forests, mangroves, wetlands, peatlands and seagrass beds.

Therefore, a key mitigation strategy is to recover these ecosystems that have been degraded, damaged or destroyed[2].

[1]CBD. Quick guide to the Aichi Biodiversity Targets, pressures on vulnerable ecosystems reduced. Available at: https://www.cbd.int/doc/strategic-plan/targets/T10-quick-guide-en.pdf Accessed on 11/16/2016.

[2]CBD. Quick guide to the Aichi Biodiversity Targets, ecosystems restored and resilience enhanced. Available at: https://www.cbd.int/doc/strategic-plan/targets/T15-quick-guide-en.pdf Accessed on 11/16/2016.

4.4.4 Biological Invasions

Nature of the driver, its recent status and trend, and factors that influence its intensity

Invasive alien species have gone from scientific curiosity to a real societal concern due to their ecological, social, and economic impacts (Mack et al., 2000). Invasive plants and animals cause changes in the composition and function of ecosystems, affecting biodiversity, ecosystem services, and human welfare. invasive alien species have become a major component of global change and pose a serious threat to local and global biodiversity (Hobbs, 2000; Mack et al., 2000; Vilà & Ibañez, 2011).

For a species to become an invasive species, it must successfully transit three distinct stages, often called the "invasion process" (Blackburn et al., 2014; Canning-Clode, 2015). The first stage of this process is the "transport phase" where individuals of a species are transported (intentionally or unintentionally) from their native range and released outside their native range. These individuals are termed "non-native" (synonymous term with the terms "non-indigenous", "exotic", and "alien"). Second, these individuals may establish a viable self-sustaining population ("establishment phase") and become "naturalized" species in the new environment. In the third and final stage, a naturalized non-native population might increase in abundance and expand its geographic range ("spread phase"), with the potential to alter the environment in which they have become established, causing ecological and economic harm ("impact phase") and becoming what is considered an "invasive species". This report uses the definition of invasive alien species of the CBD (see Deliverable 3b on invasive alien species), which defines the term (https://www.cbd.int/invasive/terms.shtml) as "plants, animals, pathogens and other organisms that are non-native to an ecosystem, and which may cause economic or environmental harm or adversely affect human health. In particular, they impact adversely upon biodiversity, including decline or elimination of native species - through competition, predation, or transmission of pathogens - and the disruption of local ecosystems and ecosystem functions."

Invasive alien species as drivers and passengers of global change

Unlike other drivers of biodiversity, biological invasions are considered both drivers and passengers of human-driven global change (MacDougall & Turkington, 2005). Biological invasions are by definition caused by the human movement of species and their magnitudes are highly associated with the intensity of changes caused by human activities (Mack & Lonsdale, 2001). Some invasive alien species may be considered passengers of global change because they only persist in an ecosystem through continued human disturbance (e.g. some European weeds associated to roadsides; Seipel et al., 2011). However, many invasive alien species also cause substantial alterations to biodiversity and ecosystem function (e.g. plants increase fire regimes or top-predators). Thus, estimating and forecasting the effects of invasive alien species on biodiversity and ecosystem services has an additional layer of complexity compared to other drivers of global change.

Invasive alien species may act synergistically with each other or with other forces of global change (e.g. climate and land use change) to produce more intense consequences for biodiversity and NCP (Sala et al., 2000; Newbold et al., 2015). Land use changes have long been recognized as a main promoter of invasive alien species across taxa (Hobbs, 2000). Changes in the dominant cover type cause shifts in species composition creating important opportunities for invasive alien species that are well adapted to human disturbances (Didham et al., 2007). From tropical to cold environments, land use changes are associated with roads and other human corridors, which are the main route for dispersal of invasive plants and animals (Seipel et al., 2012). In the last two decades, climate change has been shown to promote invasive alien species by disrupting ecosystems, but also by changing conditions so that they are more suitable to the invader than to the native community (Bellard et al., 2012).

The movement of species by humans and its successful naturalization has increased exponentially in the last two centuries (Seebens et al., 2017). In the Americas, the onset of biological invasions is marked by the arrival of Europeans in the 1500s, which resulted in the massive introduction of non-native species, and the reduction of the natural biogeographical barriers of a continent that had been isolated for thousands of years (i.e. since last glaciation). The influx of non-native species caused by European colonization is still visible today as most invasive alien species in Mediterranean and Temperate regions of the continent are from Eurasia. For example, naturalized plants in Chile and California are mostly Eurasian species (Jimenez et al., 2008). Increase in trade and connectivity, in the last two centuries, and especially since the 1900s, have facilitated the arrival of non-native species from other continents including Australia, Asia and Africa (Jimenez et al., 2008; Van Kleunen et al., 2015).

The introduction of new non-native species into the Americas is expected to continue with increasing trade and transportation by land, sea and air, increasing biological invasions and their potential impacts on biodiversity and NCP in the Americas (Early et al., 2016). Furthermore, the consequences of recent additions of non-native species to the Americas may not yet be visible because it takes time for species to reach high population numbers and wide distributions to cause detectable ecological or economic impacts (i.e. "invasion debt", Essl et al., 2011).

Significant knowledge gaps of invasive species in the Americas exist (Pysek et al., 2008). While countries such as the USA and Canada have been leaders in recording and studying invasive species, most countries in the Americas have only recently directed efforts to record invasive alien species and their impacts (Pysek et al., 2008; Pauchard et al., 2010). Auspiciously, national inventories of invasive species and research on invasive species and their impacts is now being promoted across the Americas to reduce this knowledge gap (e.g. Mexico, Chile, Brazil, Argentina; Zenni et al., 2017).

In the following sections, we review some of the most relevant impacts causes by invasive species in each of the regions of the Americas and their main ecosystem units and we emphasize their role as drivers of changes in biodiversity and their interactions with other drivers of global change.

North America

North America is one of the most invaded regions of the world and one of the most studied in terms of the numbers and impacts of biological invasions (Jeschke & Strayer, 2005; Pysek et al., 2009). Since the 1500s, trade and land use change drivers, in this region, has consistently promoted the establishment of some of the most damaging plant and animal invasive alien species (Stohlgren et al., 2006). The advance of the chestnut rust that decimated the natural populations of the American chestnut (*Castanea dentata*) exemplifies the magnitude of the species, community and ecosystem level impacts of biological invasions in North America (Jacobs et al., 2013). Reductions of plant diversity caused by direct competition between native and non-native plants have been extensively reported in grasslands of North America (Vilà et al., 2003). Plant invasions have also caused enormous changes in ecosystems processes such as hydrological and fire regimes. For example, cheatgrass (*Bromus tectorum*) invasion in arid grasslands has resulted in more frequent and more damaging fires (Pawlak et al., 2014). In addition, some of the most well-known examples of animal invasions have occurred in North America. Vertebrate predators such as rats (e.g. *Rattus rattus*), carps (*Cyprinidae* spp.), and snakes (e.g. *Python bivittatus*) have substantially altered native animal populations driving some to near extinction (Dorcas et al., 2012). Non-native insects, such as ants and mosquitoes, have had a large impact on human well-being (Juliano & Lounibos, 2005).

Tundra and mountain grasslands show relatively low number of plant invasions because of the climatic barrier and the relatively low levels of human disturbances (Pauchard et al., 2009; Bellard et al., 2013). However, some species, mostly European ruderals, are widely distributed in mountains and alpine ecosystems (Alexander et al., 2016). Because of the low abundance and frequency, few impacts have been reported of these plant invasions. Similarly, other taxa invasions have been scarcely reported in these cold ecosystems, partly because the lack of surveys and studies. Climate change and increasing human pressure will likely change this scenario, also causing unexpected shifts in native species distributions (Pauchard et al., 2016).

Boreal and temperate forests and woodlands pose a significant barrier to plant invasions because of the high competition for light (Martin et al., 2008). Thus, most ruderal plant invaders, which invade roadsides and disturbed areas, are not able to succeed in the forest understory (Martin et al., 2008). Nonetheless, in eastern North America, species that are shade tolerant are now entering forested areas. For example, garlic mustard (*Alliara petiolata*) is now occupying deciduous forests generating monospecific patches and displacing native understory species (Kurtz & Hansen, 2014). On the other hand, these forests have been heavily impacted by animals and pathogens. For example, earthworms are now considered a major driver of change in temperate forests (Bohlen et al., 2004). Invasive insects, such wooly adelgids, have devastated forests in eastern USA, having broad range impacts including indirect impacts on fish in streams due to loss of shading (Ellison et al., 2005).

Temperate grasslands have suffered extreme transformations in North America, being replaced by agricultural lands or when maintained, have gone intense grazing pressure and heavy disturbance (e.g. plowing). Thus, the remaining grasslands in North America are being intensively affected by plant invasions. Ruderal species of Eurasian origin such as *Centaurea* spp. *Euphorbia* spp. and *Bromus* spp. have replaced native grasses and herbs across the North American grasslands (Stohlgren et al., 1999). Their impacts not only include changes in plant cover but also long-term shifts in soil processes, food webs and fire regimes (Simberloff et al., 2013).

Mediterranean forests, woodlands and scrub in North America are one of the hotspots for invasive plant species (Seabloom et al., 2006). The high level of trade and human-caused disturbance in this area, and the close climatic match with Mediterranean Europe are responsible for the high levels of invasive plant species (Seabloom et al., 2006). Some of these species have caused irreversible ecosystem change by replacing native species and creating a positive feedback with fire (see example of *Bromus* above). Fungi pathogens have also affected the health of these ecosystems (e.g. Oak Death, Rizzo & Gargelotto, 2003).

Drylands and deserts in North America have been invaded by non-native grasses, shrubs and trees. Invasive species capable of standing desert conditions have thrived in the shrubland and grassland vegetation competing directly for water with native species and creating a continuous fuel layer that promotes more intense and larger fires (Brooks & Chambers, 2011). *Tamarix* invasion in riparian corridors have displaced native riparian vegetation and altered ecosystem structure (Merritt & Poff, 2010).

Wetlands in North America show the highest levels of plant invasions due to the intense purposeful or accidental introductions of aquatic plants (Batzer & Baldwin, 2012). Many of these invasive aquatic plants have profound environmental and economic costs such as *Eichornia crassipes*, *Phragmites australis*, *Lythrum salicaria,* and *Egeria densa*.

In freshwater systems, the zebra mussel (*Dreissena polymorpha*), originally (1988) affected the Great Lakes area, but has now spread to all of the large navigable rivers in the eastern USA, extending along the Illinois River to the Mississippi River and into the Caribbean (Benson et al*.,* 2017). Human activities are important vectors of transport of this species between aquatic systems (Johnson & Padilla, 1996), which is notorious for their biofouling capabilities by colonizing different human aquatic infrastructure (e.g. water supplies for hydroelectric and nuclear power plants, public water plants and other industrial facilities), causing high economic costs and having profound effects on the aquatic ecosystems they invade (Griffiths et al*.,* 1991; Pimmentel et al., 2000; Bykova et al*.,* 2006; Ward & Ricciardi, 2007). Invasive fish, such as round goby (*Neogobius melanostomus*) or Asian carp (*Cyprinus carpio*), have also impacted freshwater ecosystems and reduced native fish populations (Kolar et al., 2007; Freedman et al., 2012; Kornis et al., 2013).

In coastal ecosystems of North America, 298 non-indigenous species of invertebrates and algae have been recorded as naturalized (Ruiz et al*.,* 2000). Most non-indigenous species are crustaceans and molluscs and have resulted from ballast water, inferring that source regions of non-indigenous species differ among coasts, corresponding to local and global trade patterns. Further, at least 100 species of non-indigenous fish and 200 species of non-indigenous vascular plants are known to be established within North America coastal area (Ruiz et al*.,* 2000). North American mangroves are considered to be protected from invasions due to the harsh hydrological and edaphic conditions in which they grow. However, there is an increasing number of invasive species being reported in mangrove ecosystems associated to anthropogenic and natural disturbances (Lugo, 1998), including the Brazilian pepper *Schinus terebinthifolius* raddi (Anacardiaceae) in Florida (Ferriter, 1997) and the Indo-Pacific lionfish *Pterois volitans* (Linnaeus, 1758) (Scorpaenidae) from North Carolina to Caribbean (Barbour et al., 2010).

Urban sprawl in North America is a major driver of landscape change and cities are a contributing source of invasive species to the surrounding rural or natural matrix. Ornamental plants, pets and pests have higher chances to adapt and invade natural systems as the propagule pressure (i.e. events of introduction) increases. Insects such as the argentine ants have also exploited human disturbances around cities (Holway et al., 2002).

Mesoamerica and the Caribbean

As of 2006, Mexico's National Commission for the Knowledge and Use of Biodiversity identified at least 800 invasive species in Mexico, including 665 plants, 77 fishes, 2 amphibians, 8 reptiles, 30 birds and six mammals, with significant ecological and economic impacts.

Buffel grass (*Pennisetum ciliare*) has invaded many of the drylands in Mexico (Marshall et al., 2012) after being introduced in the 1970s into Sonora from the USA to bolster the cattle industry (Cox et al., 1988; De La Barrera & Castellanos, 2007; Franklin et al., 2006). From 1973 to 2000, Buffel grass pastures in Mexico increased from 7,700 hectares to 140,000 hectares (Franklin et al., 2006). It is estimated to cover 53% of Sonora and up to 12% of Mexico overall (Arriaga et al., 2004). Buffel grass invasion can devastate local ecosystems by increasing wildfire regimes, soil erosion rates, ground surface temperatures and supply of vital resources to surrounding life forms, compromising biodiversity (D'antonio & Vitousek, 1992). Buffel grass is also present in Central American countries like Nicaragua, El Salvador, Honduras, and, Panama (Global Biodiversity Information Facility, 2011).

The southern Yucatán peninsular region is the largest continuous expanse of tropical forests remaining in Central America and Mexico, it has been identified as a hotspot of forest and biotic diversity loss (Achard et al., 1998). Bracken fern (*Pteridium aquilinum* (L.) Kuhn) invasion have spread under agriculture cultivation (Schneider, 2006). Frequent fires and land clearance for agriculture have facilitated the replacement of secondary vegetation with bracken fern (Schneider & Nelun Fernando, 2010). The feral pig (*Sus scrofa*), from the same species as the European wild pig, has invaded the Coco's Island Marine and Land Conservation Area, a national park in the Costa Rican Pacific (Hernández et al., 2002). Because of their rooting activity, these animals alter approximately 20% of the island surface each year, leading up to eight times the erosion in the affected area. These animals also eat fruits, earthworms, roots, stems and leaves, reducing the layer of organic material in leaf litter and plant cover.

Invasive insects are also wide spread throughout Mesoamerica. The Mediterranean fruit fly *(Ceratitis capilata)*, heads the list of invasive alien species of economic importance in the Mesoamerican region, and is considered a genuine pest affecting all Central American countries. This insect, which entered the region in 1955, attacks fruit and fills it with worms. As a result, some fruit exports from Central America to the USA were suspended. Fruit trade with Europe and Japan has also been affected.

In freshwater ecosystems, African cichlid fish, *Oreochromis* spp., were accidentally introduced in Lake Chichancanab two decades ago, in the central Yucatán Peninsula in Mexico, causing change in the native fish diversity and in the transmission of endemic trematodes to the piscivorous birds (Strecker, 2006). Nile tilapia *(Oreochromis niloticus)* is currently found in the Apoyo, Nicaragua and Managua lakes (Nicaragua), Caño Negro Wildlife Refuge, and Lake Arenal (Costa Rica). This species has resulted in a decline of approximately 80% in the biomass of native cichlidic fish in Lake Nicaragua and has displaced native fish in Caño Negro due to increased competition and predation.

Introduced fish species often result in alteration of food webs. Two exotic fish, common carp (*Cyprinus carpio*) and tilapia (*Oreochromis niloticus*), were introduced for aquaculture more than 20 years ago into the Xochimilco wetlands, Mexico City and now dominate the system in terms of biomass and numbers. Over this period, wild populations of the microendemic axolotl salamander (*Ambystoma mexicanum*) have been dramatically reduced (Zambrano et al., 2010).

In the Mexican Caribbean, the Indo-Pacific lionfish (*Pterois volitans*) has become a species of great concern because of their predatory habits and rapid proliferation throughout the Mesoamerican Barrier Reef, the second largest continuous reef system in the world (Valdez-Moreno et al., 2012). Having few predators, this invasive predatory fish can greatly reduce native fish biomass and is a threat to the marine environment throughout the region (Green et al., 2012) (Box 4.21).

The seaweed flora of California, USA and Baja California, Mexico is highly diverse and is now being threatened by invasive species that are largely introduced unintentionally. Most of the 29 non-native seaweed species that have been recorded, originated in Asia and have been introduced within the last 30 years. The vectors that bring these plants or their propagules to the California and Baja California coasts (international shipping (e.g. ballast water) and shellfish aquaculture) may have not changed drastically in the last decades, but the conditions for the establishment of non-native species seem to have improved. Climate change, including the frequency and severity of El Niño Southern Oscillation events, may be responsible for creating space, diminishing competition, and permitting the persistence and spread of non-native species (Miller et al., 2011; Kaplanis & Smith, 2016).

In the Caribbean islands, humans have introduced many plant and animal species (Kairo et al., 2003; Rojas & Acevedo, 2015; van der Burg et al., 2012; Jenkins et al., 2014), and non-native species have often become ubiquitous there. Caribbean terrestrial ecosystems have been heavily invaded by plants and animals. For example, forest inventories of various Caribbean islands, based on plots or remote sensing, have found that forests dominated by non-native tree species are extensive (Chinea & Helmer, 2003; Brandeis et al., 2009; Helmer et al., 2012), although some of these new tree communities may have a beneficial role. For example, early successional species often dominate and catalyze understory colonization by native tree species (Parrotta, 1992; Parrotta et al., 1997; Wolf & van Bloem, 2012), or when legumes or nutrient-rich leaves attract insects that provide more forage for insectivorous birds. Shade-tolerant non-native species, however, can be common in forest understories (Brown et al., 2006) and could permanently change species composition by effectively competing with late successional native species.

The marabú, (*Dichrostachys cinerea L.*), an invasive Fabaceae, has invaded almost 800,000 hectares of Cuba's forests (Hernández et al., 2002). This thorny bush grows in forests and abandoned agricultural fields, leaving infested areas unproductive. Nowadays, marabú has become Cuba's primary problem with respect to invasive alien species, in terms of both economic and environmental impacts. Environmentally, the most serious damage is inflicted on fields (livestock) and on forest plantations. Lands invaded by marabú remain unusable and thorny, impassable for livestock and human beings. In its juvenile state, marabú is practically impenetrable since it forms extremely dense thickets up to five meters high. In the case of forest plantations, this invasive bush is highly expensive to control. The country spends millions of USA dollars a year to combat this species, but its great capacity for reproducing through seeds, trunks and roots makes it very difficult to eliminate. More information on invasive species in Cuba is presented in supplementary material: Box 4.22.

Many of the problems of Mesoamerican invaders in ocean ecosystems are repeated throughout the Caribbean. The Indo-Pacific lionfish (*Pterois volitans* and *P. miles*) waslikely introduced in the USA state of Florida through aquarium releases, and has quickly spread to all tropical and subtropical coastal waters of the western Atlantic Ocean and Caribbean Sea (Schofield, 2010). In fact, this species may be the most damaging marine fish invasion to date (Hixon et al., 2016) (Supplementary material, Box 4.21 and Mesoamerica section above).

South America

South America, due to its relative isolation, was until recently, considered to be relatively less affected by biological invasions (Speziale et al., 2012). However, evidence has shown that biological invasions are occurring in ecosystems that were considered protected, such as the Andes mountains (Pauchard et al., 2009), the Amazon basin (Silvério et al., 2013), and the Patagonian south Atlantic coast (Oresanz et al., 2002). These large and diverse ecosystems harbor a number of invasive species, including some of the world's worst invaders (Speziale et al., 2012). The mongoose (*Herpestes javanicus*), introduced as a predator of rats and snakes, spread preying on endemic fauna and transmitting rabies and leptospirosis (Ziller et al., 2005). Other introduced species act as ecosystem engineers, transforming and threatening complete ecosystems (Speziale et al., 2012), as well as changing their services (e.g. beavers *Castor canadienses*; Anderson et al., 2006 and Box 4.23 in supplementary material and *Limnoperna fortunei,* Boltovskoy et al. 2015 and Box 4.24, in supplementary material). Crop species with important commercial value, have also become invasive. Pines (Pinaceae family) for example, used widely as a forestry cultivar, are invasive in both temperate and tropical regions because they have been planted extensively and have biological attributes that promote their invasiveness (Pauchard et al., 2015).

Invasive species in South America come from all continents, although Europe is a major donor of invasive species, especially for plants (Van Kleunen et al., 2015). Undoubtedly, the number of new introductions is increasing annually because of intensified trade and transport routes which is diversifying the source of invasions (Speziale et al., 2012). Harbors, roads, airports, and cities are major sources for the entry of new species. For example, big metropolitan areas such as Sao Paulo, Santiago, or Buenos Aires are centers for the introduction of new invaders (e.g. Masi et al., 2010). Also, the increase human footprint in the landscape (section 4.4.1), and the introduction of new species for cultivation, is increasing the chances for new invasions.

Invasive species can also come from within the same country. For example, introduced marmosets in southeastern Brazil have been reported as a potential threat to local biodiversity. Marmosets compete with other primate species and birds for resources (Lyra-Neves et al., 2007), depredate birds and eggs (Galetti et al., 2009), hybridize with conspecifics (Begotti & Landesmann, 2008), and transport new pathogens (Sales et al., 2010).

Tropical and subtropical humid and dry forests are one of the most extensive ecosystems in South America and are being impacted by several species that mostly originated from other tropical areas in Asia and Africa. While many tropical forests appear to be substantially free of invasive species, some species are able to invade mainland forest ecosystems where canopy structure is naturally open, rainforests are fragmented or disturbed, or forests are exploited for crops or timber (Denslow & DeWalt, 2008). In addition, fires reportedly interact with grass invasion through a positive feedback cycle, causing a decline in tree cover, facilitating grass invasions, and increasing the likelihood of future fires. In the tropical dry forests of Bolivia, grasses have invaded the forest where disturbance coincides with seed dispersal by motor vehicles involved in logging activities (Veldman & Putz, 2010). In the tropical and subtropical forests of Brazil, some of the most invasive plants known by their ability to outcompete native species, are *Artocarpus heterophyllus* and *Hedychium coronarium* in tropical ombrophilous forest, *Hovenia dulcis* in subtropical ombrophilous forest and subtropical semi-deciduous forest, *Pinus taeda* and *Pinus elliottii* in subtropical ombrophilous forest and steppe, and *Tecoma stans* in tropical and subtropical semi-deciduous forest (Zenni & Ziller, 2011). Tropical forest biotas are susceptible to taxonomic homogenization (i.e. increasing levels of similarity and reduce biotic differentiation) due to the increase of some generalist invaders that replace more specialized native species (e.g. the Atlantic forest of northeast Brazil, Lôbo et al., 2011).

Mediterranean forests, woodlands and scrub are one of the invasion hotspots of South America because of their high human footprint and climatic similarities with biomes in Europe and North America. Ruderal agricultural weeds, native to the Mediterranean region of Europe, are widely distributed and invade natural ecosystems, increasing homogenization and affecting ecosystem dynamics (e.g. intensifying fire regimes) (Jimenez et al., 2008; Castro et al., 2005). Animal invasions are also affecting the processes of this ecosystem. For example, the European rabbit (*Oryctolagus cuniculus)* exerts a profound herbivore pressure in the Mediterranean scrub (Camus et al., 2008, Iriarte et al., 2005).

Tropical savannas and grasslands have been heavily affected by invasive African grasses. African grasses are used for pasture improvement, recovery of degraded areas, and slope cover along highway and railway embankments (Reis et al., 2003; Martins, 2006). Invasive grasses have been identified as a degradation driver of Colombian wetlands (Ricaurte et al., 2014), while in the Cerrado biome of Brazil, they constitute a serious problem because they invade open areas (Pivello, 2014). Molasses grass (*Melinis minutiflora* P. Beauv.) accumulates more biomass than do most other species of the herbaceous stratum vegetation native to the Cerrado (Rossi et al., 2014). The effect of invasive grass cover is especially high on the Cerrado-specialist species, whose proportion has consistently declined with increasing invasive dominance. Thus, invasive grasses reduce the floristic uniqueness of pristine vegetation physiognomies (Almeida-Neto et al., 2010). In savannas and grasslands, invasive trees have become problematic. For example, the invasion by *Pinus elliottii* is one of the most serious threats to the remaining native Cerrado vegetation causing biodiversity losses (Abreu & Durigan, 2011).

Temperate grasslands in South America are highly threatened by invasive species because of their long history of agriculture and livestock usage that has causedinvasive species to become widely distributed. For example, in the Argentina pampas, introduced forage grasses, such as *Festuca arundinacea* and *Lolium multiflorum*, and weedy forbs such as *Carduus acanthoides*, heavily dominate secondary grasslands on former arable fields (Tognetti et al., 2010) and invade native grassland remnants grazed by cattle (Perelman et al., 2007; Tognetti & Chaneton, 2015).

Drylands and deserts of South America show relatively low numbers of invasive plant species (Fuentes et al., 2013). However, some succulent plant invaders such as *Mesembryanthemum* spp are invading desert islands in northern Chile (Madrigal-González et al., 2013) and invasive animals such as rabbits and feral goats are having a strong effect on vegetation and overall ecosystem dynamics (Meserve et al., 2016).

Temperate and boreal forests and woodlands have a relatively low area in South America (see Chapter 3 for more details). However, they show a high level of endemism and represent the most southern forests in the world (Rozzi et al., 2008). These forests are being invaded by herbs, shrubs, and trees mostly brought to Chile for agricultural use, erosion control, forestry, and ornamental use (Pauchard et al., 2015). For example, *Acacia* and *Pinus* species arewidely used in forestry, and are a problem in the temperate forests of south-central Chile where they outcompete native vegetation and increase fire regimes (Fuentes-Ramirez et al., 2011; Le Maitre et al., 2011; Langdon et al., 2010; Cobar-Carranza et al., 2015). Several invasive vertebrates are also invading these forests (e.g. wild boar, red deer, mink; Iriarte et al. 2005), with the most damaging being the North American beaver, which has decimated forests (i.e. cutting and flooding) in the southern tip of the continent (Anderson et al., 2006; see Box 4.23, supplementary material).

Although tundra and mountain grasslands are considered less invaded than lowland ecosystems, recent evidence shows that there is an increasing number of invasive plant species being established at higher elevations in the Andes (Pauchard et al., 2009; Alexander et al., 2016). Species, such as *Taraxacum officinale*, may have important impacts on pollination, reaching high elevations beyond the treeline (Muñoz et al., 2005). As climate warming progresses, there is a greater chance of higher latitude and elevation plant invasions (Lembrecht et al., 2015).

Freshwater ecosystems are suffering strong transformation due to invasive species. For example, *Limnoperna fortunei,* commonly known as golden mussel, have invaded major rivers of the Río de la Plata basin and associated tributary basins via ballast water. Because of the ecological effects caused in aquatic ecosystems and expenses incurred in industrial infrastructure, it is considered a high priority aquatic invasive species to be addressed at the regional level (Boltovskoy, 2015) (see Box 4.24, supplementary material). *Lithobates catesbeianus* native frog from the southeast of USA has colonized more than 75% of South America where it has been reported to be a highly effective predator, competitor, and vector of amphibian diseases (Laufer et al., 2017). Climate change may have a potential synergistic effect on the invasion of this frog throughout the Atlantic forest biodiversity hotspot (Nori et al., 2011). The microalgae *Didymosphenia geminata*, an invasive freshwater benthic diatom native to rivers of the Circumboreal region of Europe, was reported in Argentinean and Chilean freshwater rivers. Thisalgae has been characterized as one of the most aggressive invasions in recent history, resulting in severe ecological and economic impacts due to the velocity of expansion and the number of rivers affected (Jaramillo et al., 2015).

In marine ecosystems of South America during the decades 1990-2000, ballast water, biofouling, and aquaculture vectors moved several coastal marine species from distant biogeographic provinces (e.g. Indo-

Pacific and Asia) to coastal environments of America (Orensanz et al., 2002; Salles & Correa da Silva Luz de Souza, 2004). These species have become invasive, resulting in negative effects on ecosystem services provided by various aquatic ecosystems. The golden mussel (*Limnoperna fortunei*) in the Río de la Plata basin have modified the provision of freshwater services (potable and industrial uses) (Boltovskoy, 2015) and food services (malacological resources) due to effects of predation on native malacofauna by *Rapana venosa* in the Río de la Plata (Brugnoliet al., 2014) (Box 4.25, supplementary material). Finally, the Indo-Pacific lionfish (*Pterois volitans* and *P. miles*) affects food (fisheries) and cultural (tourism, recreation: diving) services at the north coast of South America (Colombia, Venezuela) due to predation of indigenous fish fauna of megadiverse coastal marine ecosystems (e.g. coral reefs) (Box 4.21, supplementary material). However, because of euryhaline and eurythermal features of this species, their expansion has not been constrained by the Amazon-Orinoco plume (Luizet al., 2013), being recently reported in the southeastern coast of Brazil (Ferreira et al., 2015).

In the marine environments off Patagonian shelf and Chilean Pacific coast, a series of biological invasions including algae, mollusks, hydroids, bryozoans, ascidiaceans, and crustaceans (at least 41 invasive alien species) occurred with severe consequences for local biodiversity with economic impact (Bigattiet al., 2008; Orensanz et al., 2002; Penchaszadeh et al., 2005). *Undaria pinnatifida* is a successful invasive seaweed widespread along the coast of Patagonia. Its presence is associated with a dramatic decrease in species richness and diversity of native seaweeds (Casas et al., 2004; Irigoyen et al., 2011). For Brazilian shelves, Lopes et al. (2009) have compiled information on the threat of invasive species. Currently, 66 invasive species have been recorded for the marine environment in Brazil from the following groups: phytoplankton (3), macroalgae (10), zooplankton (10), zoobenthos (38), fish (4), and pelagic bacteria (1) with different ecological and economics impacts in marine Brazilian ecosystems (Lopes et al., 2009).

4.4.5 Overexploitation

Nature of the driver, its recent status and trend, and factors that influence its intensity

Overharvesting, or overexploitation, occurs when humans extract more of a natural resource than can be replaced naturally. This unsustainable practice threatens biodiversity and can degrade ecosystem services by reducing species populations below natural self-sustaining levels and disrupting ecosystem functions and species interactions. Overharvesting can happen in hunting, fishing, logging, groundwater mining, overgrazing, or the collection of wild plants and animals for medicine, decoration or for the pet trade. Harvested species are used as food, building and other industrial materials, medicines, fibers for clothing, ornamental items, as well as in other social and cultural aspects.

Growing human populations, rising incomes, consumer demand, expanding markets, and improved technology all contribute to overharvesting. Individuals, communities or corporations that have open and unregulated access to public goods like forests, aquifers, fisheries, and grazing lands can overexploit a shared resource to maximize short-term profits until it eventually becomes unavailable for the whole (Hardin, 1968). Harvesting natural resources is an essential part of livelihoods and economies of all worldviews. When people act in their own self-interests, they tend to consume as much of a scarce resource as possible, leading to overharvesting and in some cases extinction or resource depletion. Early examples include, the Steller's sea cow (*Hydrodamalisgigas*), once found throughout the Bering Sea, was hunted into extinction within 27 years of discovery for its meat, fat, and hide; and the passenger pigeon (*Ectopistesmigratorius*), once considered the most abundant bird species on the planet, was hunted to extinction over a few decades throughout North America (Bucher, 1992). There are many examples linking extinction to joint effects of harvesting and habitat change as extensive areas in eastern North America were converted to agriculture and urbanization.

Overexploitation of species often leads to cascading effects with sometimes irreversible impacts on trophic-level functions and can negatively affect the structure, dynamics, or quality of an ecosystem. This is particularly true if a habitat loses an apex predator which can result in a dramatic increase in the population of a prey species. In turn, the unchecked prey can overexploit their own food resources to their own demise and impact other species (Frank et al., 2005; Borrvall & Ebenman, 2006; Heithaus et al., 2008). Fishing down the food chain, where larger predatory fish, such as cod, tuna, and grouper, are targeted first, followed by smaller fish in the food chain, causes trophic level dysfunction (Pauly et al., 1998). Some species require a sufficient density of individuals to reproduce and when reduced to smaller populations, they become vulnerable, suffering from lower genetic diversity and an increased likelihood of being eliminated by natural disasters or diseases (Lacy, 2000).

When a species in not able to reproduce faster than it is harvested, it becomes increasingly rare which can drive its price higher in the illegal wildlife trade. This in turn, increases the incentive to extract which can cause the population to eventually collapse (Brook et al., 2008). Wildlife trade poses the challenge of separating legal from illegal trade (Broad et al., 2003) and governments can deter such illegal trade by measures such as policies that strengthen enforcement, curb the demand, and expand international cooperation to stop the illegal trade.

Many countries are responding by implementing strategies that mitigate or avoid negative impacts of overharvesting such as strengthening management regulations and enforcement, providing incentives to fishermen, foresters and others to become long term stewards of the resource, through the establishment of effectively managed protected areas and no-take zones, as well as strengthening institutions and regulations to eliminate illegal wildlife trade and put in place sound practices to regulate legal exports/imports of vulnerable species. Tenure rights and other means of co-management are also ways in which local communities can have more say over their natural resources and long-term conservation. For example, territorial user rights in fisheries, such as those set up in Chile for the small scale artisanal fishing sector, provide incentives to maximize economic benefits and encourage greater stewardship of the resource to local communities. Individual transferable quotas or other catch share strategies can also be applied to larger scale fisheries to prevent collapses and restore declining fisheries although critics point to them being exclusionary and involve trade-offs, such as changes in fleet capacity, employment, and aggregation of fishery shares (Costello et al., 2008). However, many States have implemented measures to manage the potentially disruptive effects of individual transferable quotas. These practices should be accompanied with investments in sustainable alternative livelihoods and wide-spread education that can inspire conservation of local habitats and species and promotes the ability of local institutions to implement and sustain conservation programs.

Terrestrial

Overharvesting of terrestrial species and resources is often driven by the pursuit of quick short-term gains without regard to the long-term effects. Illegal logging, for example, can include overharvesting of large tracks of forests or the selling of rare wood species. It is pervasive throughout Mesoamerica and South America and impacts many different stakeholders and communities that rely on timber for their livelihoods (Richards et al., 2003). Capital-endowed actors as well as poor forest dwellers may drive overharvesting, albeit for different reasons (Pokorny et al., 2016). Poor governance, corruption, and rampant demands for space to carry out socio-economic activities (e.g. cattle grazing) contribute to the problem. Curbing this problem is difficult. For example, in the Amazon region, timber companies, as well as illegal harvesters, seeking to adopt sustainable practices face challenges such as high investment costs, large transport distances, lack of capacity, and resources to implement environmental regulations (Pokorny et al., 2016). The pattern of deforestation can be exacerbated once timber companies provide road access and infrastructure to previously intact areas, allowing small landholders to continue to overharvest with often no management or enforcement.

Unsustainable hunting and collection of species driven by market demand is another contributing factor of overharvesting. The animal diversity that Central and South America holds and the limited enforcement of wildlife trading laws creates a magnet for wildlife traffickers and the lucrative exotic pet trade. However, the sustainability level of harvest for the majority of species is unknown. Birds are the most trafficked for pets, but reptiles like iguanas, snakes, and turtles are highly valued as pets as well as for their skin, shells, and eggs (Shirey et al., 2013). Amphibians, scorpions, spiders, and insects are also collected (Ripple et al., 2015; Broad et al., 2003). Products are often sold for ornaments and furnishings include coral, turtle and mollusk shells, and reptile skins (Shirey et al., 2013), many other products are sold as traditional “medicine” especially to Asian countries. In addition to the pet trade, there is an estimated eight million people in South America that rely regularly on bushmeat as a source of protein in their diets. While this represents only

1.4% to 2.2% of the total continental population, these people are likely to be some of the poorest in the region (Wilkie & Godoy, 2001). The distinction between subsistence and commercial use is often unclear and more research is needed on subsistence vs non-subsistence harvesting and how much of subsistence harvesting is optional but local (i.e. they have other sources but choose to eat bushmeat when available).

Plants and fungi provide people with food, medicine, building materials, and as raw materials for making other products. Some species are highly valued for their beauty or medicinal value. Thousands of medicinal and aromatic plants that are collected in the Americas are used in the international trade and are valued at over $1.3 billion (Lange, 1998). Many species of ornamental plants, like flowers, orchids, tree ferns, bromeliads, cycads, palms, and cacti, are commercially overexploited in both legal and illegal markets. For example, orchids throughout North and South America are one of the best-selling in the legal horticultural trade but are also traded illegally and make up 70% of all species listed by the Convention on the International Trade in Endangered Species (CITES). Research conducted by Hinsley et al. (2015) in the Americas indicates that two key consumer groups purchasing rare plants are either serious hobbyists, who prefer rare species, or mass market buyers whose preferences are based on aesthetic attributes.

Freshwater resources

The Americas show wide variation in overexploitation of surface and groundwater resources. Large portions of South and Central America, Canada, and Alaska are relatively water secure, while the western half of the USA, nearly all of Mexico and the Caribbean, and coastal portions of South America all experience seasonal and dry year water depletion (Brauman et al., 2015). Climate change is expected to exacerbate water shortages in many parts of the Americas (UNEP, 2010; IPCC, 2014a).

Surface water depletion can have visible impacts as streams dry up, but groundwater depletion is no less serious and can have longer-term consequences. Sustained groundwater pumping can lead to drying up of wells, reduction of water in streams and lakes, deterioration of water quality, increased pumping costs, and land subsidence (Konikow, 2013). Depletion of ground water in the USA is a serious problem as aquifers provide drinking water for about half the total population and nearly all rural population as well as providing over 50 billion gallons per day for agricultural needs. The cumulative depletion of groundwater in the USA between 1900 and 2008 was about 1,000 km^3—equivalent to about twice the water volume of Lake Erie (Konikow, 2013).

Irrigation is by far the largest source of water consumption globally and in the Americas. Domestic use is the second largest consumer in North and Central America, while in South America livestock production is slightly higher (Brauman et al., 2015). Overharvest of water in general has implications not only for human communities, both in terms of water quality and quantity, but also for aquatic and even terrestrial species whose life cycles are adapted to natural flow regimes (Poff et al., 1997).

Impacts to species from overexploitation of water largely track where that overexploitation is greatest. An analysis of species listed as extinct through vulnerable in the IUCN Red List finds that only 5% of assessed species associated with South American inland wetlands are threatened by water abstraction, whereas the numbers rise to 17% in Mesoamerica and 32% in North America (IUCN Red List, 2016). These numbers should be interpreted with caution, given that comprehensive species assessments are lacking for much of Latin America. Overharvesting of freshwater species in the Americas is considered in general less of a threat to biodiversity and ecosystem services than the degradation and alteration of the habitats in which those species live (Welcomme et al., 2011). However, overharvest can combine with those impacts, which include but are not limited to changes to hydrology, connectivity, and water quality, to impair species and services further (Allan et al., 2005).

Freshwater species

Globally, most inland fisheries are comprised of small-scale fishers, whose catches are underreported by as much as a factor of two (Coates, 1995; Mills et al., 2011). Even with underreporting the level of fisheries exploitation in Latin America has been judged to be lower than in Africa and Asia; however, specific fisheries show signs of overharvest (Welcomme et al., 2011; Muller-Karger et al., 2017). For instance, overfishing of valuable freshwater fish species and turtles has been documented in tributaries of the Amazon (Alho et al., 2015). In general, national governments have underinvested in monitoring inland fisheries because those fisheries are assumed to be of low value. Consequently, the range of threats to those fisheries, including overexploitation, are poorly documented (FAO Committee on Fisheries, 2014).

Cascading effects of freshwater overharvesting are numerous and include the phenomenon of "fishing down", in which exploitation leads to depletion of high-value, large-bodied fish species and the consequent reduction of mean body size of harvested species (Welcomme, 1999; Pauly & Palomares, 2005). This has been documented in the Amazon and elsewhere, with implications for food web structure, water quality, and nutrient cycles; these changes, in turn, have been implicated in the ecological extinction of species like manatees (Castello et al., 2013; Castello et al., 2015).

Marine

The most significant driver of overharvesting in the marine environment is fishing. With population growth and incomes rising, the demand for seafood continues to grow for both human consumption and feed for livestock and aquaculture. Fishing remains a key source of food and employment for millions of people in the Americas and a significant factor in regional economies. About 2.4 million fishers and 10% of the world's motorized fishing vessels are in the Americas (FAO, 2016c), landing 18.5 million metric tons of seafood in 2013 (FAO, 2016b). From 1961 to 2013, the per capita annual seafood consumption in the Americas rose 26% from 7.9 to 10.7 kg (FAO, 2016a). Different large marine ecosystems of the Americas (Sherman et al., 2005; Sherman & Hamukuaya, 2016) show different top-down pressures and strong regional differences in oceanographic properties which shape the diversity and abundance of the catch within these regions (Muller-Karger et al., 2017). The adoption of more efficient fishing technologies has also contributed to the rapid depletion of fish stocks, the endangerment of charismatic marine species, and the loss and degradation of marine habitats. An estimated 34% of the assessed stocks in geographic areas surrounding the Americas (FAO regions 67, 77, 87, 21, 31, and 41) were deemed overexploited in 2009 (FAO, 2011). However, the adoption of fishing technologies has been documented to have positive effects as well, such as much lower bycatches and less habitat impacts.

Invertebrates like squids, shrimps, lobsters, crabs, oysters, and sea cucumbers account for roughly 20% – 3.7 million tons – of the seafood caught in the Americas in 2013 (FishStatJ, 2016). Many of these fisheries and their habitats are at risk from overexploitation. For example, 85% of the world's oyster reefs have disappeared since the late 19th century, largely due to habitat degradation, with many formerly prolific reefs rendered "functionally extinct." Overharvesting is the main cause of oyster reef loss, however direct habitat loss is also a significant problem caused by commercial ship traffic, pollution, and aquaculture, among others. Other invertebrates, like seas cucumbers, have plummeted across the Americas due to high demand from Asian markets.

A consequence of fishing is the unintended catch of fish and other marine organisms, also known as bycatch. Hundreds of thousands of sea turtles, seabirds, whales, dolphins, and porpoises die globally each year from being caught as bycatch in regular fishing operations. As many as 200,000 loggerhead turtles and 50,000 critically endangered leatherbacks were killed as bycatch on longlines in 2000 (Lewison et al., 2004); longlining is also estimated to kill between 160,000 to 320,000 seabirds annually (Anderson et al., 2011). Several studies report that the use of bycatch reduction devices can successfully reduce bycatch species while maintaining target catch rates (Favaro & Côté, 2013; Pelc et al., 2015). The vaquita, a small porpoise in Mexico's Gulf of California, have been driven towards extinction as they are killed after getting entangled in gillnets used to catch shrimp and other fish; only 30 are estimated to remain (Morell, 2017).

Sharks and rays are severely overfished globally, with an estimated 97 million caught each year either in direct target fisheries or as bycatch in other fisheries (Clarke et al., 2013). One-quarter of the 1,041 assessed sharks, rays, and chimaeras are threatened under the IUCN Red List criteria due to overfishing, however nearly half are considered too data-deficient to be classified. Many shark species are pelagic and migratory—some with a circumglobal distribution across temperate and tropical oceans—meaning that overharvesting of sharks in the Americas contribute to a global problem. Only 23 sharks and rays had been listed under CITES up to 2016, when an additional 13 species of sharks and rays were listed. Trade restrictions on listed species and bans on shark finning have increased during the last decade, however they have not significantly reduced shark mortality or risk to threatened species (Davidson et al., 2016). Some countries, such as The Bahamas, have implemented a national ban on the harvest of sharks, protecting more than 40 species of sharks.

Additional drivers of overharvesting in the America's marine environment include hunting, aquarium trade, medicinal use, and entanglement in fishing and marine gears. Turtles, narwhals, and corals are harvested for ornamental and jewellery making, and live fish, corals, and invertebrates are harvested for the aquarium and pet trade. Some species like sea horses are also targeted for traditional medicinal use primarily in Asian markets. Direct harvest of non-fish species, like seals, otters and whales, has seen a reduction since the peak of these industries almost a century ago, but some of these species continue to be harvested, particularly in Canada. An estimated 308,000 whales and dolphins die each year from the consequences of entanglement in fishing gear, laceration, infection, and starvation) (International Whaling Commission https://iwc.int/entanglement).

North America

Terrestrial

An example of an overharvested plant in North America is American ginseng (*Panax quinquefolius*), a species found in the temperate eastern forests and is prized for its medicinal properties that has received increased scientific and commercial attention. Due to the plant's very specialized growing environment and demand in the commercial market, it has started to reach an endangered status in some areas (McGraw et al., 2010). Acts, such as the Endangered Species Act, have succeeded in reducing the harvest of rare species, preventing the extinction of hundreds of additional American wildlife species since 1973 (Adkins, 2016).

Freshwater

While loss of spawning beds and pollution contributed, overfishing in the Great Lakes is a good example of inland surface water overharvesting that has caused whitefish, walleye, and sturgeon populations to decline. Recreational fisheries are also poorly documented, by and large; in Canada, however, the collapse of four inland fisheries has been associated with recreational fishing (Cooke & Cowx, 2004). Within coastal and inland rivers, the well-documented decline of Pacific salmon and other anadromous fish species as a result of overfishing, dams, and other threats has led to cascading effects including the loss of nutrient inputs to terrestrial systems (Marcarelli et al., 2014). Four native freshwater turtle species (*Chelydra serpentine*, *Apalone ferox*, *Apalone mutica*, and *Apalone spinifera*) now require increased protection driven by trade to Asia (USFW, 2014).

Marine

In North America, fishing remains the primary driver of overharvesting in the marine environment. In the USA, fish stocks are generally well-managed, at least at the federal level. For the 233 stocks with known status only 16% are overharvested, while overharvesting occurs in only 9% of the 313 stocks with known status (NOAA, 2016). Several overharvested species have been well-documented, like the collapse of the Atlantic cod of the Scotian bank, which provides a classic example of overharvesting that resulted in the closure of a 9,600 square miles area in 1994 (Frank et al., 2005). There has been a reduction in the direct harvest of marine mammals that have historically been overharvested, like seals, otters, and whales since the peak of these industries almost a century ago. For example, sea otter (*Enhydra lutris*) hunts peaked in the middle of the 1800s when the species was almost driven to extinction by the fur trade. Sea otters were listed under the U.S. Endangered Species Act in 1977 and designated endangered in Canada in 1978, and most of their historical range has been reoccupied, but their numbers are still considered low in some areas (Bodkin, 2014). For oyster reefs, overharvesting remains a serious problem as about three-quarters of the world's remaining wild oyster reefs are found in just five locations in North America, however only in one of these regions — the Gulf of Mexico — are oyster populations deemed relatively healthy as of 2011 (Beck et al., 2011).

Several policies have reduced or eliminated the harvesting of selected species like the U.S. Marine Mammal Protection Act of 1972 that established a moratorium on the taking of marine mammals in USA waters and the USA passed the Endangered Species Act (1973) that restricts harvests of critically imperilled species. In

1973, CITES was established to ensure that international trade of animals and plants does not threaten their survival in the wild. Canada and the USA often use allocation of fishing rights and use of protected areas to manage fisheries in federal waters, with agencies establishing quotas using robust stock assessments and monitoring programs. Examples of overharvesting in North America Artic and Greenland are presented in Box 4.14.

Box 4.14. Overharvesting in North America Artic and Greenland

Several fisheries studies in northeastern Canada and Greenland observe species overharvesting which can lead to cascading effects and modification of food webs (Jørgensen et al., 2014; Shelton & Morgan, 2014; Munden, 2013). Overexploited fish species include Atlantic cod (*Gadus morhua*), Atlantic halibut (*Hippoglossus hippoglossus*), redfish (*Sebastes mentella*), Atlantic wolffish (*Anarhichas lupus*), starry ray (*Rujuradiuta*), and American plaice (*Hippoglossoides platessoides*). Deep-sea fish species are particularly vulnerable to overexploitation as they mature late and have a low fecundity and slow growth rate (Jorgensen et al., 2014). Barkley (2015) reports two key datasets to develop sustainable harvest levels for Greenland halibut (*Reinhardtius hippoglossoide*s) in the Canadian Arctic and understanding the stock connectivity between inshore and offshore environments as well as examining capture induced stress metabolites in Greenland halibut caught in a trawl and Greenland sharks (*Somniosus microcephalus*) caught as bycatch on bottom longlines.

Mortality of non-target species, or bycatch, is a fisheries management problem that can be solved with innovative fishing gear and practices. Traditional fishing gears, like trawls, not only contribute to bycatch, but can greatly modify marine habitat. FAO (2016c) reports that 35% of landings are bycatch with at least 8% being thrown back into the sea. In Newfoundland, Munden (2013) found that impacts of bycatch and habitat alteration can be mitigated through gear modification. She found that a modified shrimp trawl can reduce contact area by 39% while increasing shrimp harvesting by 23%. A change in the type of gangions can lead to a significant reduction in shark bycatch without negatively impacting commercial catches of turbot (*Scophthalmus maximus*). In Davis Strait, West Greenland, one of the world's largest cold-water shrimp fisheries, with an annual catch of about 80,000 tons, bottom trawls have excessively modified bottom habitats and community structures (UNEP, 2004). Jorgensen et al. (2014) studied nine bycatch species from bottom-trawl surveys of Greenland halibut over a 24-year period and found that four populations showed a significant reduction in mean weight of individuals that was significantly correlated with increases in fishing effort.

Mesoamerica

Terrestrial

Mesoamerica provides an important corridor for many Neotropical migrant bird species and home to rare and charismatic species like the scarlet parrot, ocelot, beaded lizard, river turtle, and the iconic jaguar that are threatened by the illegal pet trade. Butterflies, reptile leather, shark fin are also popular items on the black market. In the tropical dry and humid forests, several valuable tree species like mahogany and black rosewood are increasingly in demand and being cut and smuggled into markets in India and China by organized crime (Dudley et al., 2014; Blaser et al., 2015). In 2016, rosewood species have been included in CITES. The southern border of the USA is also a hot zone for wildlife smuggling based on the nearly 50,000 illegal shipments of wildlife and wildlife products that were seized at ports of entry from 2005 through 2014. This included nearly 55,000 live animals and three million pounds of wildlife products (Defenders of Wildlife, 2016)

Marine

While most high migratory species are assessed and well-managed through multinational efforts in Mesoamerica, many coastal fish stocks are considered to be overfished or declining (FAO, 2011). Examples of locally overfished species groups throughout Mesoamerica include crabs, sea-spiders, and shrimp, as well as various demersal fish (croakers, snappers, groupers) that form a large portion of the bycatch from shrimp fisheries (FAO, 2011). The vaquita have also become overfished to endangerment in recent years after becoming entangled bycatch in gillnets set for the totoaba, a large white fish (Morell, 2017). The overharvesting of sea turtles continues to be a problem as all seven species of sea turtles are threatened by the sale of meat, jewelry, and leather products. Their eggs are sold on a thriving Central American market as a male aphrodisiac. Heavy exploitation of sea turtles in the Mexican and Caribbean regions began in the 15th century. In the 1970s, sea turtles were added to Appendix I of CITES, banning commercial trade between member states. Despite CITES and U.S. Endangered Species Act listing, sea turtles are still declining. Turtles also die in huge numbers entangled in the nets of fishers. Another species threatened by trade and illegal harvest is Mexico's totoaba, an endangered fish endemic to the Gulf of California. Totoaba are valued for its swim bladders, used to make a specialty soup, and individual fish can be sold for $10,000 to $20,000 apiece in the Asian market (Neme, 2016). Sea cucumbers also remain overexploited throughout Mesoamerica, driven by lucrative export markets to Asian countries (Purcell et al., 2013). Effective fisheries management regulations and capacity are lacking in many parts of Mesoamerica. In cases where management systems do exist, they are often jeopardized by data deficiencies, a lack of enforcement and monitoring, and corruption. Lack of effective management has led to de facto open access and overfishing.

Caribbean

Marine

According to the FAO, the Caribbean Sea (FAO area 31) has the highest proportion of overfished stocks in the world, about 54% in 2009 (FAO, 2011). Long-term catch data suggest that fish catches in the Caribbean increased by about 800% since 1950, and have been declining since 2001. Conclusions about the recent declines in fish landings as indicators of the status of fish stocks can only be made with very low certainty as the fish landings data comprise multiple fish species across many trophic levels, data sources have changed over the years, and landings from artisanal fishers are thought to be unreported. However, it is likely that the declining trend in fish landings indicate decreases in the size of fish stocks across the region (Agard et al., 2007).

Overfishing is affecting virtually all Atlantic coral reefs and particularly in the Caribbean, with almost 70% of reefs at medium or high risk (Burke et al., 2011). Atlantic reefs have some of the lowest recorded fish biomass measures within reef habitats in the world – largely from overfishing (Burke et al., 2011; Jackson et al., 2014). While the Caribbean only supplies a small percentage of the global trade in marine ornamental species, the environmental and biological impacts of the industry are well recognized. At least 16 Caribbean countries have export markets for ornamental reef fish, with the biggest markets being the USA, the European Union, and Japan. The impacts of the ornamental reef fish industry include the overharvesting of key species, coral reef degradation associated with gear impacts and from use of cyanide and other poisons, changes in the ecology of the reefs due to focused collection of specific trophic groups like herbivores, and loss of biodiversity due to removal of rare species (Bruckner, 2005). While less than 1% of the stony corals that have been reported to CITES database originate from the western Atlantic reefs, the USA and most Caribbean nations have prohibited the trade of stony corals. Hundreds of other genera of invertebrates, including echinoderms, sponges, molluscs and crustaceans are also collected and exported from the western Atlantic, primarily for the aquarium trade (Bruckner, 2005). An additional case study on queen conch in the wider Caribbean is explained in Box 4.15.

Box 4.15. Overharvesting of queen conch in the wider Caribbean

With a life span of up to 40 years, the queen conch (*Strombus gigas*) is a unique marine mollusc found in tropical waters throughout the wider Caribbean, Bermuda and the Gulf of Mexico. Its shell is emblematic of the oceans it inhabits with many cultures referring to conch shells as a "megaphone" for hearing the ocean's sound. In addition to the ornamental use of its shell, conch shells are used in jewelry making.The meat is consumed throughout the Caribbean and exported as a seafood product to the USA, France and other countries. Live queen conch are also sold in the aquarium trade. Because of its slow growth and density requirements to reproduce, queen conch are easily overharvested and the Americas have plenty of cases where this overharvesting is evident (Appeldoorn et al., 2011).

In the USA, Florida's queen conch fishery collapsed in the 1970s and today both recreational and commercial harvests of queen conch are prohibited in the State. Demand for queen conch however remains high. Since the 1980s, commercial catch has increased in response to international market demand, especially from the USA, which imported approximately 80% of the annual queen conch catch in 2004 (Paris et al., 2008). Regulatory measures to manage queen conch stocks in the region vary considerably throughout the Caribbean (Berg & Olsen, 1989; Chakalall & Cochrane, 1997). Some countries have minimum size restrictions on harvested conchs; others have closed seasons, harvest quotas, gear restrictions, spatial closures, or a combination of these; however in management response at all levels, from regional to local, has been slow in tackling overexploitation (Appeldoorn et al., 2011). In 1992, queen conch became the first large-scale fisheries product regulated under Appendix II of CITES. Appendix II includes species that are not necessarily threatened with extinction, but unless trade is strictly controlled, may become extinct. Despite CITES listing, conservation actions and management policies, few countries report substantial recovery of queen conch populations, which may be due to reduced densities that limit reproduction (Stoner & Ray, 1996; Stoner, 1997; Paris et al., 2008). More science, monitoring and management action will be required to put conch on the path to recovery and it will take time, resources and political will to achieve sustainability of this emblematic species.

South America

Terrestrial

South America is home to a multitude of species that are highly prized for the pet trade, bush meat, and traditional medicines. Many of these species are harvested by indigenous peoples and sold to traffickers. The wildlife trade affects endangered and valuable birds, mammals, reptiles and amphibians, fish, and rare trees and plants. Some bird species, like the blue-throated macaw (*Ara glaucogularis*) are prized for their brilliant color and command a high dollar price on the illegal pet trade. Estimates of annual bushmeat consumption for the Brazilian Amazon are estimated at 89,000 tons (Peres, 2000 in Ripple et al., 2015). In remote forest areas, eating bushmeat may be a matter of survival, being often the main (or only) source of animal protein available. When wild fish is available the role of bushmeat in people's diets may drop, thereby their consumption seems to be closely linked to both availability and/or prices (e.g. Rushton et al., 2005 in Peru; Nasi et al., 2011). As a cascade effect, a decline in one wild resource may drive up an unsustainable exploitation of the other (Brashares et al., 2004; Nasi et al., 2008 in Nasi et al., 2011). Nevertheless, for richer sectors of society, bushmeat is harvested for sports hunters and as a novelty food for tourist in high-end restaurants in the region.

Freshwater

Manatees (*Trichechus inunguis*) and giant otters (*Pteronura brasiliensis*) are the most demanded aquatic species of mammals found in wetlands with very high demand as food and leather, respectively. Caimans (black giant caiman, *Melanosuchus niger*, and spectacled caiman, *Caiman* spp.), the Orinoco crocodile (*Crocodylu sintermedius*) and river turtles (mainly the Amazon giant turtle – *Podocnemis expansa*) are under strong harvesting pressure in the wetlands. While caimans are still found in healthy and very abundant populations in more remote areas, clear of human interference, river turtles struggle to resist to very high harvest rates (Seijas et al., 2010; Turtle Conservation Coalition, 2011). In the Amazon and Pantanal, the overexploitation of large frugivorous fish may affect the dispersal of seeds within wetlands covering 15% of South America by area (Correa et al., 2015). Ornamental fish are caught in large numbers in the Amazon, and there is evidence of overharvest of species like the cardinal tetra (*Paracheirodon axelrodi*) (Begossi, 2010).

Even though Amazonian wetland forests are the most diverse in the world (Wittmann et al., 2006) and exploited for timber for many decades (Castello et al., 2013), quite a small number (N=14) of tree species were considered especially vulnerable (Ribeiro, 2007). Forest products for manufacturing and construction include timber, rattan and bamboo for furniture, plant oils and gums, dyes, resins and latex (Shirey et al., 2013). Some species, like mahogany (*Swietenia macrophylla*), are highly valued commercially for its beauty, durability, and color. It is estimated that approximately 58 million hectares (21%) of mahogany's historic range had been lost to forest conversion by 2001 (Grogan et al., 2010). Commercial exploitation has sometimes led to traditional medicines becoming unavailable to the indigenous peoples that have relied on them for centuries or millennia. The fate of remaining mahogany stocks in South America will depend on transforming current forest management practices into sustainable production systems. Given the potential costs and benefits associated with trade, the challenges suggest that a collaborative approach between agencies, nurseries, and plant collectors is needed to regulate the trade of listed plants. There is a substantial international trade and demand for products like Brazil nuts, palm hearts, pine nuts, mushrooms and spices (Shirey et al., 2013). In regulating commercial trade, policymakers and conservation biologists may want to consider potential risks and benefits of private efforts to recover species (Shirey et al., 2013). More details on overharvesting in Amazonian wetlands are presented in Box 4.16.

Box 4.16. Amazonian wetlands

In general, overexploitation of Amazonian wetland species has two types: timber species and fish species. Main reasons include strong market pressures from an increasing affluent urban population, unregulation of markets, and adoption of unsustainable techniques of extraction and/or production of resources, reduction of stocks, depletion and even extinctions. The Amazon human population is very dependent on local fisheries for their animal protein intake. Fish consumption is among the highest in the world. And almost 50% of the fished species exploited (and more than 60% of the biomass estimate of 450,000 tons produced annually) is directly related to the Amazonian wetlands, where they use either as spawning grounds or as nurseries to larval stages. As a very selective activity, this fishery exploits only a small fraction of the local fish diversity. Consequently, many stocks of the larger species exploited are already overfished, mainly in the more populated areas of the Amazonian wetlands (Junk et al., 2007). Although almost two hundred species of fish are of commercial value, fish yields are dominated by 18 to 20 species only. There was a reduction in the mean maximum body length of the main species harvested in 1895 (circa 206 cm) to the main species harvested in 2007 (circa 79 cm). From the group of species harvested in the early 19th century, three are now endangered. From the 18 species dominating yields nowadays, one is endangered and four were found to be overexploited in at least one region of the Amazon basin (Castello et al., 2013). Modern technologies allow fishermen to explore more distant places, to travel longer and further, and to catch and store a higher amount of fish biomass.

Marine

While just over 27% of assessed fish stocks on the Pacific coast of South America are considered overexploited, roughly 69% of assessed fish stocks are overfished on the Atlantic coast. Conversely, 59% of unassessed stocks on the Pacific coast of South America are estimated to be overexploited, while 53% of assessed fish stocks are estimated to be overfished on the Atlantic coast (FAO, 2011; Hilborn & Ovando, 2014).

The Humboldt Current moves cold Antarctic waters along the western coast of South America and drives upwelling of nutrient-rich water, making the coastal shelf one of the most productive marine environments in the world. Large environmental variations are known to cause large year-to-year fluctuations as well as longer-term changes in fish abundance and total production of the main exploited species (FAO, 2011). The world's largest fishery by volume, the anchoveta, is targeted mainly by Peru and Chile. Overfishing played a major role in the collapse of the anchoveta fishery in 1973, 1983, and again in 1998, however it is also recognized that environmental conditions also significantly influenced the decline (FAO, 2016). More recently, the adoption of an individual quota system for the industrial sector of the fleet and other management measures have contributed to reducing the excess industrial fishing capacity for anchoveta. The small and medium scale sector still need reforms, but the fishery is considered by fisheries scientists to be managed within sustainable limits.

Additionally, local populations of sea urchins, clams, scallops, and other shellfishes have been overexploited in some areas (FAO, 2011). As coastal stocks decline, commercial fishers continue to move further offshore in search of higher trophic-level species that are more valuable. Lack of effective fisheries management has also led to illegal, unreported, and unregulated fishing, and exploitation by foreign fleets. The bycatch of seabirds, marine mammals, and sea turtles is thought to be significant in both southwest Atlantic and southeast Pacific for gillnet and driftnet fishing gears, although there are large data gaps in the existing knowledge its extent and contribution to the overexploitation of marine species (Wiedenfeld et al., 2015).

In the Americas, incorporation of traditional values, knowledge, and social taboos within indigenous communities is increasingly being recognized as a fundamental part of effective resource management (Colding & Folke, 2001; Heyman et al., 2001; Moller et al., 2004; Fraser et al., 2006; Herrmann, 2006). Trends are towards participatory, inclusive, community-based approaches to conservation (Berkes, 2007) that provides a sense of ownership and promotes self-management. Traditional ecological knowledge within indigenous communities accumulates across multiple generations and is learned through years of observations in nature (Drew, 2005). Invaluable local insight provides a deep understanding of the critical balance to maintain ecological integrity within an environment and it fosters shared responsibilities between locals and the science community. Moller et al. (2004) suggest that by combination traditional ecological knowledge and science, insight can be gained into prey population dynamics as well sustainable wildlife harvests. By doing so, partnerships and community buy in is garnered and indigenous users develop their own adaptive management actions which are often more effective since they have greater investment in having a sustainable resource.

4.5 Direct natural drivers

Nature of the driver, its recent status, and trends and what influences its intensity

Direct natural drivers of biodiversity loss include large environmental disturbances. Effects of disturbance on biodiversity have been studied in many ecosystems (Dornelas, 2010; Vega-Rodriguez et al., 2015). The types of disturbance include everything from single tree-falls (Brokaw, 1985) to ecological catastrophes (Hughes, 1994).

Natural disturbances are caused by natural climatic, geologic, and biological fluctuations. Large, severe disturbances are often considered natural disasters, because they can threaten human life and have striking short-term effects on plant and animal populations (Lindenmayer et al., 2009). They are often event-triggered by natural hazards that overwhelm local response capacity and seriously affect the social and economic development of a region (United Nations & The World Bank, 2010).

Globally, natural hazards are classified as: geophysical (e.g. earthquake, volcano, mass movement; meteorological (short-lived/small to meso scale atmospheric processes, e.g. storms); hydrological (e.g. flood, wet mass movement, climatological (long-lived/meso to macro scale processes, e.g. extreme temperature, drought, wildfire), or biological (e.g. epidemic, insect infestation, animal stampede) (Guha et al., 2014). Biological disasters are not included in this assessment.

Sources of risk are both natural and man-made. Ecosystem structure can ameliorate “natural” hazards and disruptive natural events. For example, vegetative structure can reduce potentially catastrophic effects of storms, floods, and droughts through its storage capacity and surface resistance whilecoral reefs can reduce wave energy and protect adjacent coastlines from storm damage (de Groot et al., 2002). Forests and riparian wetlands or coastal ecosystems like vegetated dunes, mangroves, coral reefs and sea-grass, reduce exposure to natural hazards by acting as natural buffers and protective barriers that, reducing the impacts of extreme natural events like landslides, tidal waves or tsunamis (Welle et al., 2012; Rodil et al., 2015). Consequently, environmental degradation directly magnifies the risk natural hazards by destroying natural barriers, leaveing human settlements and socioeconomic activities more vulnerable.

Climate change is predicted to increase the frequency of high-intensity storms in selected ocean basins depending on the climate model. The majority of tropical hurricanes damage from climate change tends to be concentrated in North America and the Caribbean–Central American region (Mendelsohn et al., 2012). Increasing water temperatures along the Pacific coast through strong El Niño conditions and global warming can increase hurricane intensity. Although rare, more subtropical hurricanes have developed in the South

Atlantic Ocean near Brazil. Changes in global atmospheric circulation patterns accompaning La Niña are responsible for weather extremes in parts of the world that are typically opposite to the El Niño changes.

The Americas suffered from 74 natural disasters in 2013 (Guha-Sapir et al., 2014). Hydrological disasters (43.2%) and meteorological disasters (31.1%) occurred most often, followed by climatological (20.3%) and geophysical (5.4%) disasters. Globally, the Americas (22.2%) was only second after Asia (40.7%) in experiencing natural disasters in 2013. The nature of the risk, however, is different for different subregions of the Americas as presented below.

North America

North America has a vast range of natural disasters per year with hurricanes being one of the most common. The prevailing winds in the tropical latitudes of the Northern Hemisphere, where tropical hurricanes typically form, blow from east to west directing hurricanes to the eastern and southern coasts of the USA the islands of the Caribbean, Central America, and Mexico (see next sections). Hurricanes on eastern coasts can venture much further north due to the influence of warm waters of the Gulf stream. The west coast of Central America and Mexico are often affected by severe topical storms in the Pacific Ocean, or storms that cross from the Atlantic to the Pacific Ocean. Hurricanes, tornadoes, and other ecological disturbances alter structure and create periodic forest clearings. Hurricane Katrina (a category 5 storm) was the second costliest disaster, with total losses of $140 billion (in US 2010 values) (Wirtz et al., 2014). The aftermath resulted in an estimated loss of 320 million trees in Louisiana and Mississippi in 2005 (Hanson et al., 2010). Florida, in particular, is one of the most hurricane-prone areas in the USA (Leatherman & Defraene, 2006). Delphin et al. (2013) project major hurricane-related losses in two key ecosystem services over time: aboveground carbon storage and timber volume. Other ecosystem services that are at risk due to impacts of severe storms include storm protection from coral reef and mangroves, and other benefits obtained from low-lying coastal habitats. In the west coast of the USA, major landslides have been associated with El Nino events, especially in California State, mainly from intense rainfall (Godt et al., 1999).

Earthquake and volcanic events occur along plate boundaries in the west coast. Volcanic eruptions are active in the hot spot zone of Hawaii and in the North Pacific region including volcanoes in Alaska, the Aleutian Islands, and the Kamchatkan Peninsula.

Severe forest fires occur in western North America where conditions are drier. Fires are a natural and important disturbance in many temperate forests, but natural fire regime can be changed by poor forestry management, invasive species, encroachment, and by humans. In North America, fire suppression in some areas, has contributed to the decline of grizzly bear (*Ursus arctos horribilis*) numbers (Contreras et al., 1986). Fires promote and maintain many important berry-producing shrubs and forbs, which are important food source for bears, as well as providing habitat for insects and, in some cases, carrion. Some of the largest fires in the world occur in boreal forests. Fire return times in natural forests vary greatly, from 40 years in some Jack pine (*Pinus banksian*a) ecosystems in central Canada, to 300 years, depending on climate (van Wagner, 1978). Most boreal conifers and broad-leaved deciduous trees suffer high mortality even at low fire intensities, owing to canopy architecture, low foliar moisture, and thin bark (Johnson, 1992). Generally, the ability of post-fire boreal forest to regenerate is high, but frequent high intensity fires can offset this balance. Weather and climate are determinants for behavior and severity of wildfires, along with fuel properties, topography (Pyne et al., 1996), and the effects of climate variability which are apparent as summer temperatures increase and many regions experience long-term droughts. Under warm and dry conditions, a fire season becomes longer, and fires are easier to ignite and spread. In addition, the spread of annual invasive grasses has led to much larger, more frequent fires in dryland regions (e.g. Brooks & Minnich, 2006). La Niña favors slighty higher than normal temperatures in a broad area covering the southern Rockies and Great Plains, the Ohio valley, the southeast, and the mid-Atlantic States.

Mesoamerica

Mesoamerica also faces a variety of natural disasters, with 31% caused by floods, 26% by wind storms, 19% by earthquakes and 8% by volcanoes (Charveriat et al., 2000). Rainfall-induced disasters rank first among all natural disasters in Central America. In Central America and the Caribbean, storms that develop along the intertropical convergence zone and the subtropical high-pressure zone, dominate the weather. In

Mesoamerica, it is common for two or more countries to be struck by the same rainfall event. For example, Hurricane Mitch in 1998 affected the entire region, killing more than 18,000 people (Guinea Barrientos et al., 2015). In tropical semi-deciduous forest on the Yucatan Peninsula, Mexico, species richness of bees declined after hurricane Hurricane Dean (2007), with a loss of 40% of the species present beforehand, however the native bee community returned to previous species diversity levels just two months after the hurricane, probably due to the rapid recovery of the vegetation (Ramírez et al., 2016).

El Niño years are associated with intense droughts and an increase in wildfires. In Mexico, during El Niño of 1998 near to 849,632 hectares were affected for 14,445 fires (Delgadillo, 1999). While the El Niño of 2005 registered 9,709 fires in Mexico that affected 276,089 hectares (Villers & Hernández, 2007).

There is also a great deal of seismic activity in the region due to the presence of several active geologic faults within the Central America Volcanic Arc. Volcanic eruptions and earthquakes occur frequently that have resulted in the loss of lives and property and impacted natural ecosystems.

Caribbean

In the Caribbean, windstorms constitute more than half of disasters while flooding is the second most common disaster. Floods are a function of climate, hydrology, and soil characteristics and are usually associated with hurricanes and other tropical storms which generate heavy rainfall. Small Island Developing States of the Caribbean are particularly vulnerable. The region experiences regular annual losses due to natural hazard events in the order of $3 billion (Collymore, 2011). In Haiti, a devastating earthquake struck the island in 2010, killing more than 300,000 people. The human impact of the earthquake was immense primarily because it occurred in a large urban area with many poorly-constructed buildings (Zephyr, 2011). Geology and climate contribute to the prevalence of landslides in the Caribbean. Weather patterns, deforestation in some places, and increasing population density are among the major causes of landslides in the region (Holcombe et al., 2012). Droughts have also negatively affected the economic and social sectors of several Caribbean states and are often related to the El Niño Southern Oscillation. Some countries in the region, like Guyana in 1997 and Cuba between 2004-2006 and 2015-2017, experience severe droughts that direct influence biodiversity and ecosystem services. The Caribbean and eastern Central America are also prone to disturbance due to tsunamis, which have historically caused substantial loss of life and property in many countries of the region (Henson et al., 2006)

Huge and very rare catastrophes affecting entire regions are likely to remain imprinted in the structure of local biological communities for millennia (Brooks & Smith, 2001). The increasing frequency and range of natural disasters which, when coupled with the intensified vulnerability in the Caribbean, demonstrates the need for sustained regional efforts to reduce vulnerability to climatic and environmental hazards there. Given that the Caribbean coastal zones are at the heart of the tourism industry in the region, the economy and well-being of many countries is immensely vulnerable to natural disasters.

South America

In South America, between 1904 and 2011, 966 natural disasters were recorded, 735 of which of hydrometeorological nature. The most common events were floods and earthquakes corresponding to more than 55% of the calamitous occurrences in South America, however droughts and floods affected the largest number of people in the period (Nunes, 2011). El Niño events have resulted in higher rainfall in Peru, Ecuador, Argentina, Paraguay and Southern Brazil. The hydrological system in the region also contributes to flooding risk. The major drainage divide is far to the west along the crest of the Andes. West from this divide, in the mountainous regions, slopes of the riverbeds are very steep, which, in the event of storms, increases risk of flash flooding, the most dangerous types of flooding.

Landslides are also common in the region due to the nature of soils and steep topography and usually occur in connection with earthquakes, volcanoes, wildfires, and floods. Andean soils are relatively young and are subject to great erosion by water and winds because of the steep gradients of much of the land. Along the Andean mountain chain, landslides produce serious damage with widespread environmental and economical effects for Andean countries (Lozano et al., 2006). Landslides may have severe and long-lasting negative effects on natural and human-dominated ecosystems, but they may also influence ecosystems in positive ways. For example, landslides play a key role in the dynamics of mountainscapes and creating suitable habitat patches for some species (Restrepo et al., 2009).

With a current total of 204, South America has more active volcanoes than any other region of the world. The volcanic eruption of Puyehue-Cordón Caulle volcanic complex in Chile in 2011 dispersed about 100 million tons of pyroclastic materials. Impacts included changes in the reproduction and the body condition of a population of a lizard population (Boretto et al., 2014), increased mortality of honeybees (*Apis mellifera*) (Martínez et al. 2013), and reduced availability of forage by 90% to 100% (Siffredi et al., 2011).

Seismic activity is significant along the South American portion of the Ring of Fire. Jaramillo et al. (2012) provided the first quantification of earthquake and tsunami effects on sandy beach ecosystems after Chile's 2010 Mw 8.8 earthquake which indicated that ecological responses of beach ecosystems were strongly affected by the magnitude of land-level change.

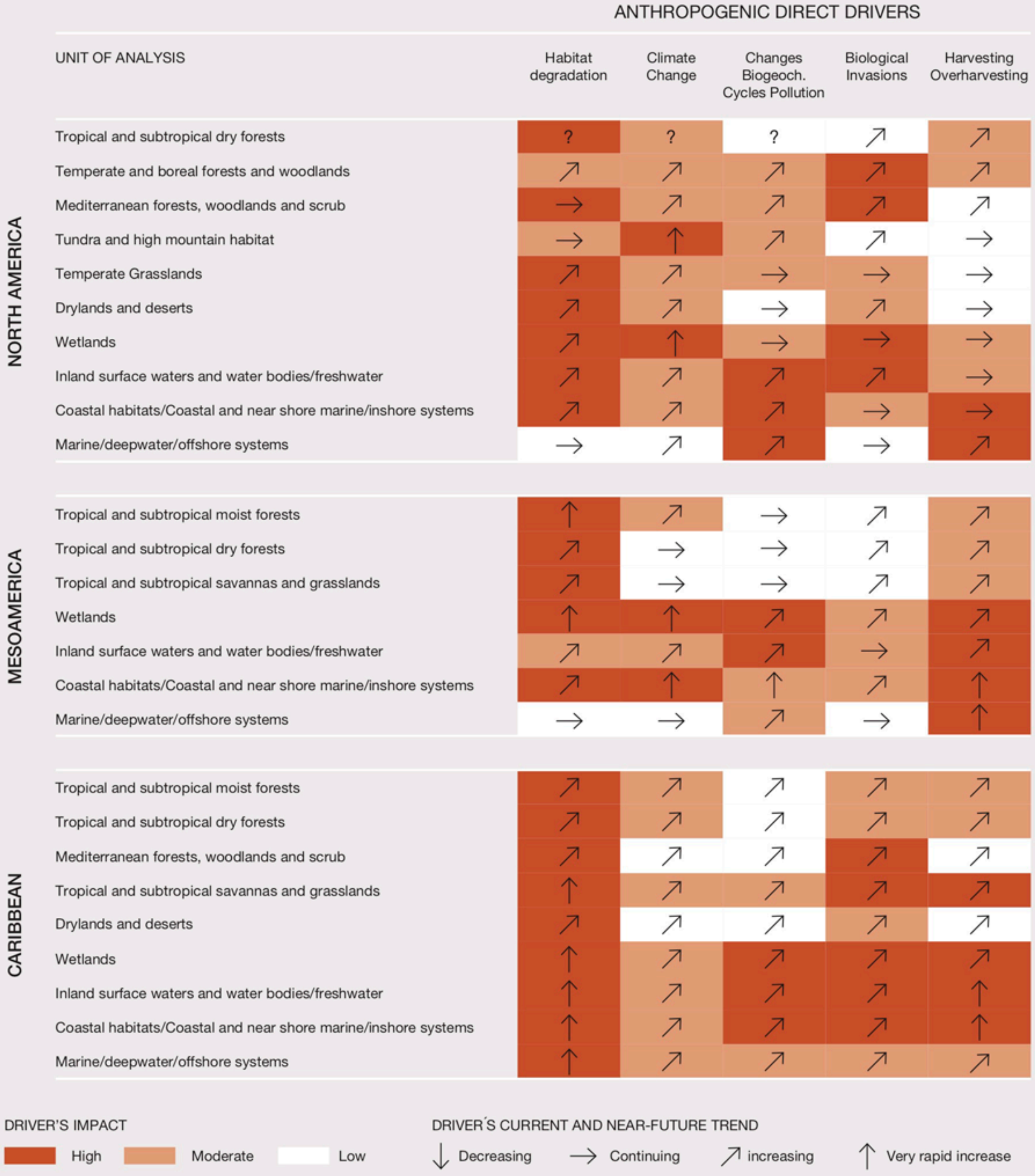
Seasonal drought occurs in climates that have well-defined annual rainy and dry seasons. However, there are important and severe drought and precipitation changes that are not seasonal and can last months to years. The arid (northeast Brazil, Mexico) and cold (south Chile) climate zones in the region have a higher propensity to drought episodes. Forest fires are associated with the dry season and drought conditions.

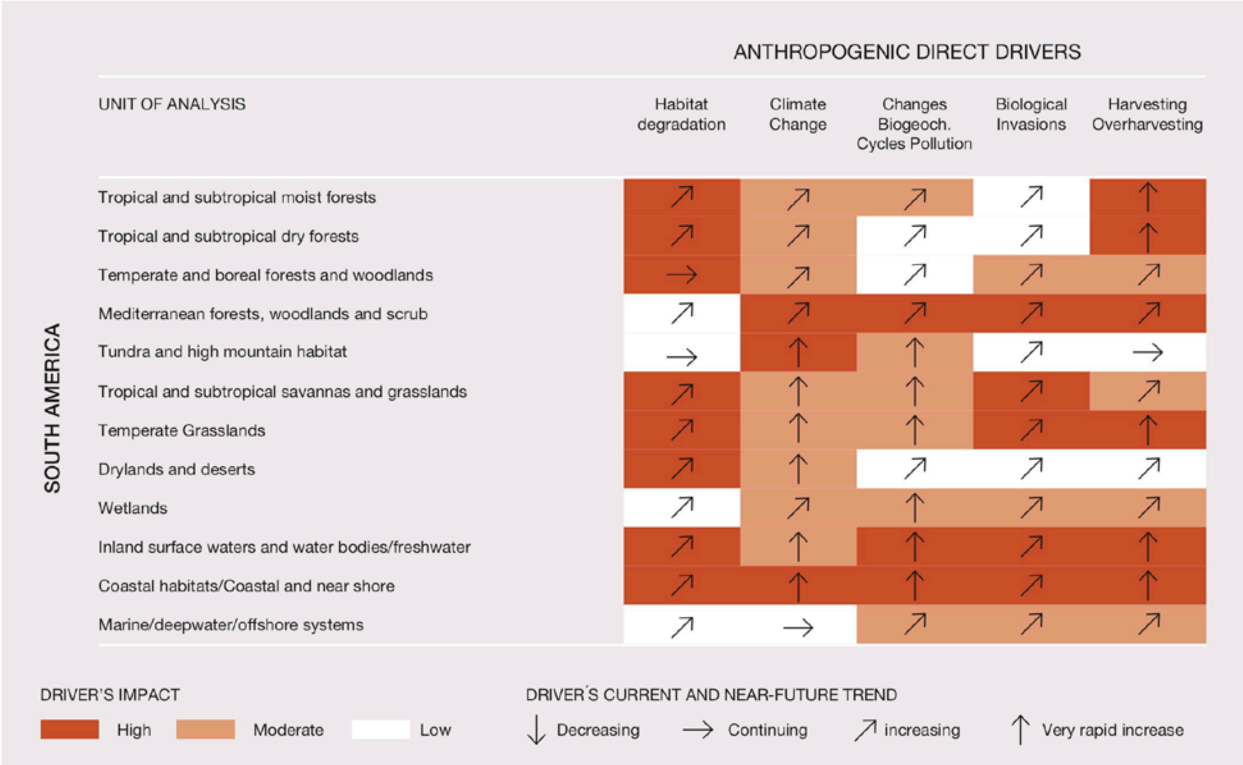
4.6 Interactions between direct drivers

Although biodiversity may also change due to natural causes (section 4.5), anthropogenic drivers dominate current change in the Americas. As presented in Figure 4.12 in all four subregions of the Americas, multiple drivers such as habitat loss and fragmentation, changes in biogeochemical cycles and pollution, climate change, overexploitation and invasive species increasingly threaten biodiversity, ecosystem services, and their benefits to society.

Figure 4 12 Relative importance of, and trends in, the impact of direct drivers on biodiversity and ecosystem services for the Americas (divided in North America, Mesoamerica, the Caribbean and South America).

Cell color indicates the impact to date of each driver on extent and condition of the units of analysis.The arrows indicate the current and near-future trend in the impact of the driver on extent and condition of the units of analysis. Change in both impacts or trends can be positive or negative. This figure is based on information synthesized from the present chapter and expert opinion. This figure presents unit of analysis-wide impacts and trends, and so may be different from those in specific sub-habitats. Source:own representation



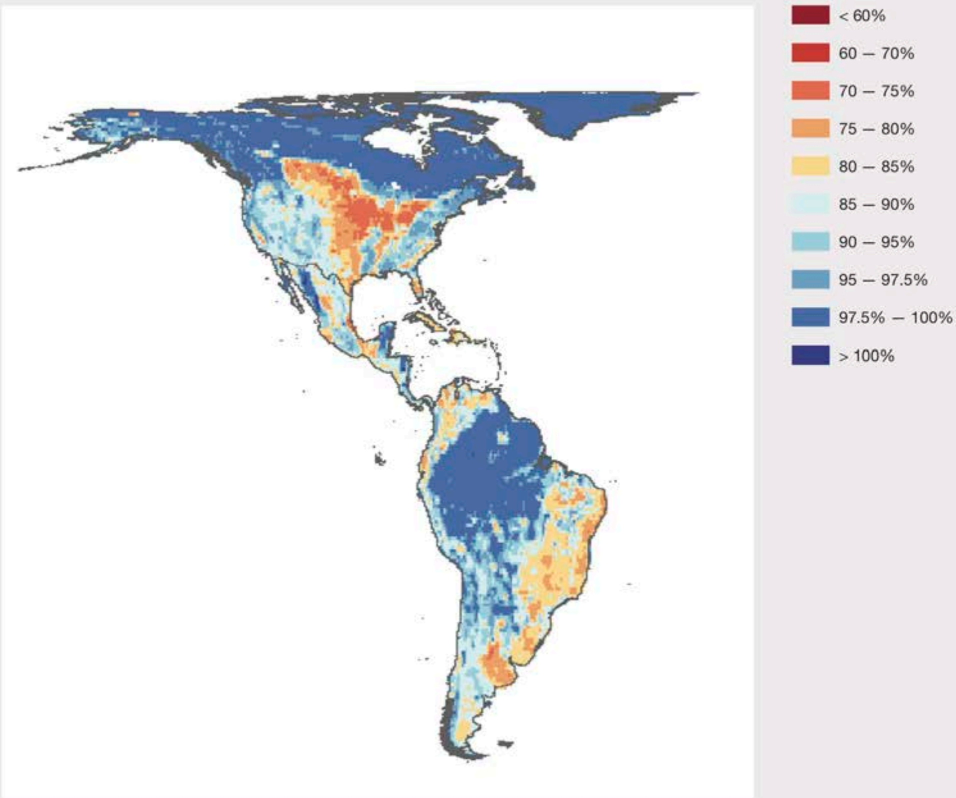


The analysis of status and trends of the different drivers indicates that habitat degradation has been the largest threat to freshwater, marine, and terrestrial biodiversity in the Americas. The net change in local diversity (for both species richness and total abundance) caused by land use and related pressures by 2005 is highlighted in Figure 4.13 (Newbold et al., 2015). All four subregions showed critical areas with significant loss of biodiversity in association to habitat degradation. As presented in section 4.4.1 and further discussed in the following section, indirect drivers such as agriculture expansion, energy demand, and urbanization are linked to extensive changes in natural landscapes.

Over time, however, it is expected that the relative importance of direct drivers will change and the effects of climate change are expected to significantly increase (Alkemade et al., 2009). The importance of the drivers of biodiversity change differs across realms, with land-use change being a dominant driver in terrestrial systems, and overexploitation in marine systems, while climate change is ubiquitous across all realms (Pereira et al., 2010). A meta-analysis of 1,319 studies that quantified the effects of habitat loss on biological populations (different taxa, landscapes, land-uses, geographic locations and climate) pointed out the magnitude of these effects depends on current climatic conditions and historical rates of climate change (Mantyka-Pringle et al., 2012). Current maximum temperature was the most important determinant of habitat loss and fragmentation effects with mean precipitation change over the last 100 years of secondary importance (Mantyka-Pringle et al., 2012).

Climate change will have far-reaching impacts on biodiversity, including increasing extinction rates. Besides exposure to climate change, there are biological differences between species that may significantly increase or reduce their vulnerability. Species that are both highly vulnerable and threatened by climate change, and the regions in which they are concentrated, deserve particular conservation attention to reduce both threats and climate change adaptation interventions (Foden et al., 2013). For example, the Amazon and Mesoamerica emerge as regions of high climate change vulnerability for both birds and amphibians, due to the large overall numbers and proportions of these groups that exist there (Foden et al., 2013).

Figure 4 13 Species richness relative to an uninhabited baseline, for the year 2000.
Source: based on Leadley *et al.* (2014).



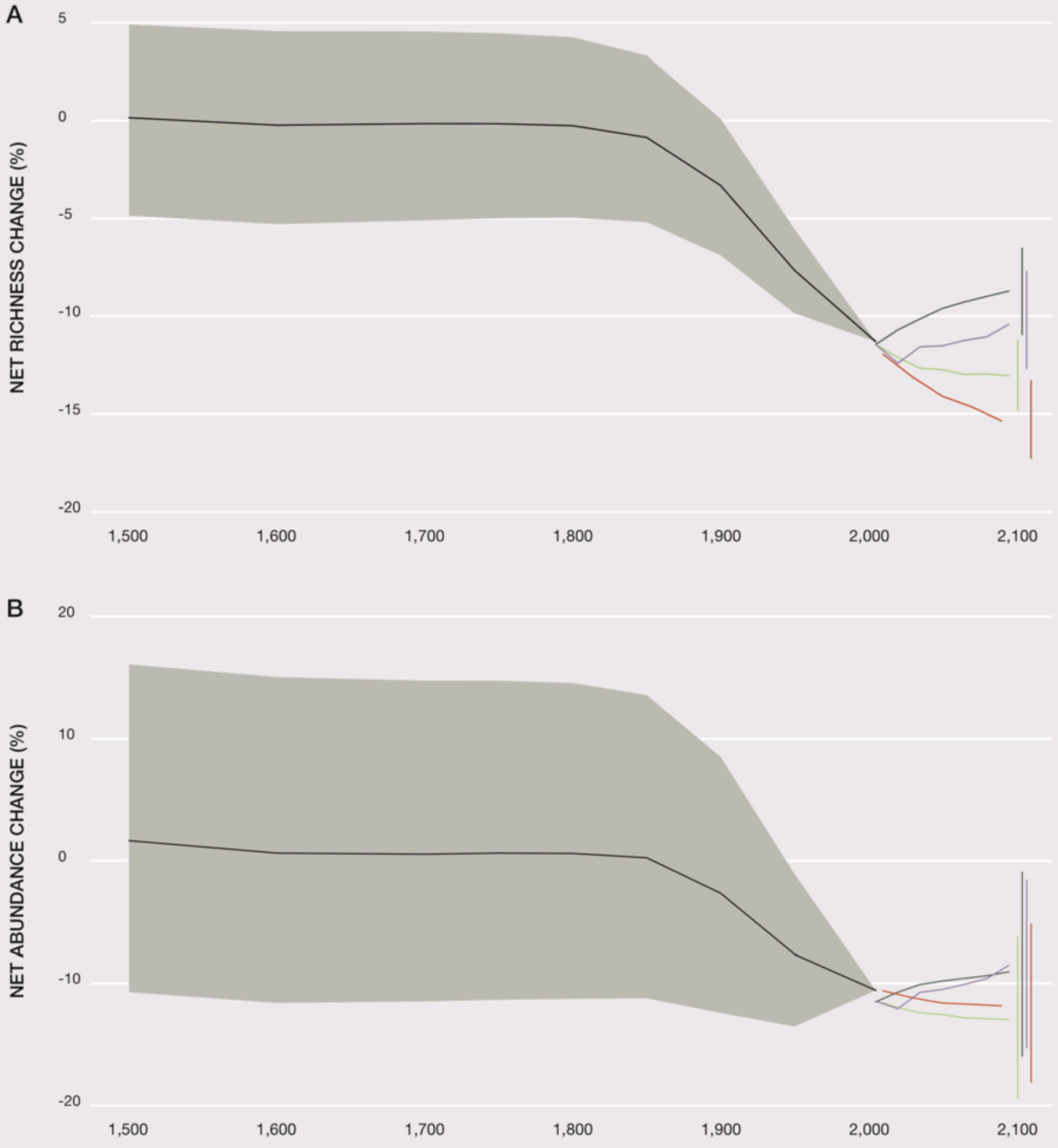
Future impacts of climate change are also related to different mitigation strategies, especially those related to land-based carbon sequestration Figure 4.4 shows historical and future estimates of net change in local diversity from 1500-2095, based on estimates of land-use intensity and human population density from the four IPCC RCP scenarios, which correspond to different intensities of global climate change (Newbold et al., 2015). Studies that addressed the interactions between land use and climate change (e.g. Oliver & Morecroft, 2014; Jantz et al., 2015) indicate the loss of natural vegetation cover generally decrease as mitigation efforts increase (RCP scenarios). The worst biodiversity outcomes arise from the scenario with the most dramatic climate change (MESSAGE 8.5) Figure 4.14 in which rapid human population growth drives widespread agricultural expansion, even though the projections omit direct climate effects on local assemblages. Recent trends in greenhouse gasses emissions most closely match this scenario (Newbold et al., 2015).

In addition, concurrent effects of climate and land use changes can further increase the already dramatic rates of biological invasions. Projections using multiple species distribution models, several global climate models, and land cover change scenarios, evaluated the vulnerability of biomes to 100 of the world's worst invasive species and highlighted the need to consider both climate and land use change when focusing on biological invasions (Bellard et al., 2013). Analysis of the future vulnerability of various biome types to these invasive alien species indicated northeastern North America as one of the three global future hotspots of invasion. Southern Brazil could be affected at a lower rate (20–40 invasive alien species) (Bellard et al., 2013).

The recognition of the interactions between direct drivers and conservation efforts implies that not only strategies focusing on a single driver might be inadequate, but also there are opportunities to align biodiversity conservation and mitigation. The cumulative and synergistic effects of drivers reinforces the need of effective adaptation strategies and policies to better safeguard protected areas under multiple drivers of change, especially since land use changes, invasives, and climate are expected to impact ecosystem function and biodiversity significantly (Hansen et al., 2014). Future trends and scenarios are developed in Chapter 5 and governance and policy options in Chapter 6.

Figure 4 14 **Historical and future estimates of net change in local diversity from 1500–2095, based on estimates of land-use, land-use intensity and human population density from the four RCP scenarios.**

Net changes in richness A, total abundance B are shown. Historical (shading) and future (error bars) uncertainty shown as 95% confidence intervals. Source: based on Newbold *et al.* (2015).



4.7 Effects of indirect drivers on direct drivers

Changes in the behaviour and values of individuals, institutions and organizations are a prerequisite for sustainable development which is a means to reduce environmental degradation and improve the quality of life within generations as well as between generations. Therefore, the identification of drivers of change, especially indirect drivers, would contribute to discerning the characteristics that need to be targeted in order to achieve sustainable development.

In the Americas, the usage and exploitation of available natural resources are expected to intensify. The indirect drivers behind this are demographic, economic, socio-political, cultural, scientific, and technological advances among others (section 4.3.). The understanding of causal dependencies between human activities and their various impacts on ecosystems is a major challenge for science and requires integration of knowledge across different ecosystem components, linking physical, chemical and biological aspects with existing and emerging anthropogenic stressors. Likewise, an effective response to these interacting threats involves a better understanding of governance systems (section 4.3) and ecological processes that affect the resilience of the biota and ecosystems including the identification of early warnings of change, tipping points and the characteristics of species, communities and ecosystems that underpin ecological resilience.

The cumulative effects of multiple stressors may not be additive but may be magnified by their interactions (synergy) and can lead to critical thresholds and transitions of ecological systems (Cotê et al., 2016). Synergistic interactions are caused by amplifying feedbacks and can provoke unpredictable “ecological surprises” that can accelerate biodiversity loss and impair the functioning of ecosystems. The conservation implications of synergies are that cascading impacts of co-occurring stressors will degrade ecosystems faster and more severely. For example, the unforeseen crash of the Peruvian anchovy populations is proposed to have resulted from the interaction between El Niño driven warming and reduced productivity, in combination with overfishing (Jackson et al., 2001).

The Americas, and in particular South American, has a major role in the global trade of products where cultivation involves deforestation and vegetation clearing in the producing countries. These products are referred to as forest and biodiversity risk commodities (Henders et al., 2015), such as beef, soybeans, maize, cotton, cocoa, coffee and timber products. There is a large potential to increase South America’s role in the trade of a number of others like palm oil and biofuels. The Americas account for the vast majority of global soy exports, for about two thirds of global maize exports and for about one third of bovine meat exports (Table 4.14 and Chapter 2, section 2.2.1). This reliance on land-based export commodities, paired with the relative abundance of arable land currently sustaining natural vegetation, clearly poses a threat to the preservation of the remaining natural areas. It has been hypothesized that in order to increase food security globally more trade liberalization is crucial, but that it would also lead to more environmental pressures in some regions across Latin America (Flachsbarth et al., 2015). The global trade network has increased enormously since 1950s in terms of the total value of exchanged goods. The technological development of means of transportations (e.g. large-scale transport of goods by airplanes, transcontinental containerships) has decreased the time necessary for transport, greatly expanded the type and value of goods transported. Increases in trading activity will cause substantial increases in invasion levels within a few decades, particularly in emerging economies (Seebens et al., 2013). These countries show most pronounced growth of naturalized plant numbers compared to countries with similar trade value increases (Seebens et al., 2013) and most of these economies coincide with regions of megadiversity (Brooks et al., 2006), rich in endemic and rare species.

The Americas experienced an early and intense urbanization process. While urbanization rates will be highest in China and India, it is in Central and South America where the largest number of species will be affected (Seto et al., 2012). Urban land-cover change threatens biodiversity and affects ecosystem productivity through loss of habitat, biomass, and carbon storage. Even relatively small decreases in habitat can cause extinction rates to rise disproportionately in already diminished and severely fragmented habitats, like the Atlantic forest hotspots in South America (Seto et al., 2012). Coastal regions and islands are particularly under pressure to increase their urban footprint. The projected urban expansion in the Caribbean islands is relatively small in total area, but they are home to a significant proportion of endemic plants and invertebrates (Chapter 3).

Energy production and agriculture are related to pollution and changes in biogeochemical cycles of major nutrients (nitrogen, carbon, phosphorus, sulfur). Atmospheric ozone occurs where emissions from fossil fuel combustion (energy utilities, industry, motor vehicle exhaust) or biomass burning interact with vapors from solvents, gasoline or vegetation. Emissions from motor vehicles and other fossil fuel combustion are also large contributors to atmospheric fine particulate matter a human health hazard. The geographic distribution of atmospheric nitrogen deposition is related to fossil fuel combustion for utilities, industry and transportation. The levels of nutrients in rivers are expected to increase in the Americas, particularly as per capita GDP, food crop, meat and milk production increase. Widespread trends in pesticide concentrations, some downward and some upward, occur in response to shifts in use patterns primarily driven by regulatory changes and introductions of new pesticides or crops, but the use of pesticides is projected to increase. Urban systems, via runoff and treated and untreated sewage, add more nutrients, sediment and organic matter to aquatic systems.

Even places with low human density are subjected to pollution from human activities. Pollution from past mining and smelting exposes wildlife to toxic metal contamination across the Americas. Lead contamination has also reached the Arctic from coal combustion and Amazonian countries are among the largest sources of mercury emissions from artisanal gold mining in the Americas. Major sources of atmospheric mercury also include fossil fuel, non-ferrous metal manufacturing, cement production, waste disposal and caustic soda production and emissions from soils, sediment, water, and biomass burning, which include re-emissions from sites that have legacy contamination issues. Toxic releases from these sites may continue due to weak environmental laws or enforcement, poor public understanding of the continuing environmental effects of these sites, and a lack of public or private funds.

The interactions between drivers presented in this chapter can be further examined using freshwater and wetland ecosystems throughout the Americas as case studies. These units of analysis appear particularly threatened in the qualitative approach presented in Figure 4.12 and their analyses can provide a means for understanding the interactions of multiple drivers with greater clarity.

Freshwater and wetland ecosystems as examples of interactions

Freshwater is an essential resource for human life and for many natural systems that support human well-being. Human alteration of rivers, lakes and wetlands has followed economic development (Revenga et al., 2005). Most freshwaters have been altered in multiple ways, and changes in any particular freshwater system usually have multiple causes. Water management is also a vast subject embracing such diverse topics as water markets, political conflict over water, connections between water and social development (Carpenter et al., 2011).

A global assessment of patterns of freshwater species diversity, threat and endemism (Collen et al., 2014), indicated that three processes predominantly threatened freshwater species: habitat loss/degradation, water pollution and over-exploitation. Of these, habitat loss/degradation was the most prevalent, affecting more than 80% of threatened species. The main indirect drivers of habitat loss and degradation were conversion to agriculture, logging, urbanization, and infrastructure development (particularly the building of dams). Dams disrupt the ecological connectivity of rivers, whereas water storage in reservoirs and release patterns affect quantity, quality, and timing of downstream flows. Consequences are influenced by interactions between different threat processes (for example, water pollution can be caused by chemical run-off from intensive agriculture or manufacturing, sedimentation by logged riparian habitat, and domestic waste water by urban expansion). On the top of these drivers climate change affects will cause impacts on freshwater and wetland ecosystems due to sea level rise, changes in precipitation, air temperature, and river discharges.

The Americas are particularly rich in terms of freshwater resources. In South America, about 30% of the planet’s freshwaters flow through the Amazon, the Parana-Río de la Plata and the Orinoco watershed. In North America, the Great Lakes shared by the USA and Canada span more than 1,200 kilometers from west to east and represent 84% of North America's surface freshwater and about 21% of the world's supply of surface freshwater. The Americas have also significant areas of wetlands. In South America, the exact size of the wetland area is not known but may comprise as much as 20% of the sub-continent, with river floodplains and intermittent interfluvial wetlands as the most prominent types (Junk, 2013). North and Central America has a combined total of 2.5 million km^2 of wetlands, with 51 % in Canada, 46 % in the USA, and the remainder in subtropical and tropical Mexico and Central America (Mitsch & Hernandez, 2013). Along the Caribbean coast and in addition to coral reefs, saltwater wetlands such as mangroves and seagrass beds are the dominant ecosystems.

Because streams, rivers, and groundwater integrate the landscape, providing a conduit for the transfer of energy and material from terrestrial habitats into freshwater systems and ultimately to the oceans, they are particularly vulnerable to environmental impacts from land use change. Wetlands are also not isolated, but are connected to their surroundings as they are often located at the transition zone between upland and open water, wetlands can be affected by activities and conditions in both terrestrial and aquatic areas. Land use influences sediment, hydrologic, and nutrient regimes, which in turn influence aquatic biota and ecological processes in freshwaters. Land use change occurs largely through human actions affected by economic incentives and regulation. These changes can have both direct and indirect effects on freshwater ecosystems - the former have immediate ecological impacts (e.g. destruction of wildlife habitats), while the latter have impacts that are transmitted via altered flow or sediment transport patterns (e.g. lower productivity due to increasing turbidity) (Palmer et al., 2002). Conversely, on many major rivers the need for hydroelectric power, flood control, and water for irrigation has led to the building of large dams that reduced the amount of sediment carried by those rivers.

North America – The Mississippi Basin

The Mississippi River watershed is the fourth largest in the world and the largest in North America at 3.2 million km^2 and includes all or parts of 31 USA states and two Canadian Provinces. Communities up and down the river use the Mississippi to obtain freshwater and to discharge their industrial and municipal waste. The Missouri River, one of the major tributaries of the basin, has had a long history of anthropogenic modification with considerable impacts on river and riparian ecology, form, and function (Skalak et al., 2013). During the 20th century, several large dam-building efforts in the basin served the needs for irrigation, flood control, navigation, and the generation of hydroelectric power. Agriculture has been the dominant land use for nearly 200 years in the Mississippi basin, and has altered the hydrologic cycle and energy budget of the region. The basin produces 92% of the USA agricultural exports, 78% of the world's exports in feed grains and soybeans, and most of the livestock and hogs produced nationally. Sixty percent of all grain exported from the USA is shipped on the Mississippi River through the Port of New Orleans and other ports in southern Louisiana.

Changes in the watershed and management practices impact the wetlands of Mississippi Delta and the Gulf of Mexico. As the Mississippi River reaches the last phase of its journey to the Gulf of Mexico in southeastern Louisiana, it enters one of the most wetland-rich regions of the world. The total amount of freshwater and saltwater wetlands has been decreasing at a rapid rate in coastal Louisiana, amounting to a total wetland loss of between 66 and 90 km^2 per year and has been attributed to both natural and artificial causes (Dunbar et al., 1992). The Mississippi River Basin accounts for 90% of the freshwater inflow to the Gulf of Mexico (Rabalais et al., 1996). Nitrate–nitrogen concentrations and fluxes from the Mississippi River Basin increased dramatically in the 20th century, particularly in the decades after 1950, when nitrogen fertilizer came into increasing use. Artificial drainage and other hydrologic changes to the landscape, atmospheric deposition of nitrates, runoff and domestic wastewater discharges from cities and suburbs, and point discharges from feedlots and other sites of intensive agricultural activity are also contributing factors to the input of nutrients into the Gulf.

South America – Río de la Plata Basin

The La Plata River Basin is one of the most important river basins of the world. Draining approximately one-fifth of the South American continent, extending over some 3.1 million km^2, and conveys water from central portions of the continent to the south-western Atlantic Ocean. The La Plata River system is recognized as among those watersheds of the world having the highest numbers of endemic fishes and birds but also the highest numbers of major dams. The La Plata Basin represents an important concentration of economic development in southern and central South America (Tucci & Clarke, 1998). Thirty-one large dams and fifty-seven large cities, each with populations in excess of 100,000 including the capital cities of Argentina, Brazil, Paraguay, and Uruguay, are to be found within this Basin. The rivers of the La Plata River Basin are subject to pressures that have modified, and can further modify the quantity and quality of their waters (Cuya et al., 2013). The consequences of these pressures are not restricted to specific countries, but are of a transboundary character. Before 1960, the Plata River Basin was almost undeveloped. The regulation of the Paraná (a large tributary of the La Plata in Brazil) for hydroelectricity has been increasing since the early 1970s. Water in reservoirs of the upper Paraná Basin currently comprises more than 70 % of the mean annual discharge at its confluence with the Paraguay River. The expansion of hydroelectric generation in the upper basin brought with it an increase in industry, agriculture, transport and settlements. These in turn increased deforestation, soil erosion, degraded water quality and reduced fisheries opportunities in both the upper and lower basins (FAO, 2016). These pressures are expected to increase in the future as the Basin countries continue to enlarge their agricultural and industrial bases, and provision of services, to improve the living standards of their increasing populations (Cuya et al., 2013). The basin has the second greatest number of planned dams in the world: 27 large dams, of which 6 are under construction. The national governments of the basin are planning a massive navigation and hydroelectric dam project (*Hidrovia*) to facilitate expansion of the export of soybean, timber, iron ore and other commodities during the dry season.

Central America and the Caribbean

Tropical rivers of Central America are highly heterogeneous systems, ranging from fast-flowing mountain torrents in areas of high relief to slow-moving rivers that meander through lowland environments. Relative to rivers in neighboring North and South America, the narrowness of the isthmus means that Central American rivers are shorter in length, carry a substantially lower volume of water as they drain smaller basins, and generally are closely connected to marine environments. Central American rivers contain hundreds of species of fishes and shrimp, including many migratory species that depend on a natural flow regime and upstream-downstream connectivity for survival. Human populations derive most water for consumptive uses from surface waters. Rivers provide a source of food, income, and building materials, serve as transportation routes, and have strong linkages to the cultural identity of rural people. Regionally, hydropower accounts for approximately 50% of net electricity generation and 42% of total installed generation capacity (Anderson, 2013). Central America has experienced a proliferation of hydropower dams in recent years, a trend that began with the construction of a few large dams in the 1980s (e.g. Arenal dam in Costa Rica, El Cajón in Honduras, and Chixoy in Guatemala), that accelerated with the privatization of electricity generation in the 1990s, and that has continued into the 21st century.

Population growth, an increase in rural electrification, and rising electricity consumption (estimated at 4.2% regionally in 2011) and reduced availability of domestic fossil fuel sources are important drivers of hydropower development in Central America. Expansion plans for the period 2012–2027 include many new hydropower developments in Central America, including large dams as well as small and medium-sized dams. Although a critical source of electricity, existing dams in Central America have been linked to declines in migratory and sensitive fish species, compromising other ecosystem services, and having negative impacts on population health and well-being. In the Caribbean, erosion, sedimentation, pollution, water nutrient enrichment, saltwater intrusion, and loss of biodiversity have been identified as the most significant factors affecting wetlands. The causes of these impacts include deforestation, tourism, urban development, industry, agriculture, damming and diversion of rivers, and dredging for navigation. In addition, natural and human enhanced phenomena such as tropical storms and hurricanes, sea level rise, and global warming also threaten these valuable ecosystems.

The challenge of matching scales: drivers, ecological and social responses

Systematic conservation planning must also ensure that not only biodiversity but also the supporting ecological processes are protected at a relevant and appropriate scale (Possingham & Wilson, 2005). Drivers interact across spatial, temporal, and organizational scales. Studies indicate that different drivers of biodiversity-ecosystem function relationships occur at small plot scales (species identities, composition) and large landscape scales (biomass, species richness) as well as in short and long temporal scales. These results imply that not all relationships and findings obtained by studies at small spatial and short temporal scales can necessarily be translated to larger or longer scales that have relevance for political decisions and conservation biology (Brose & Hillebrand, 2016). Global trends (e.g. climate change or globalization) can influence regional contexts and local ecosystem management while changes in national regulations might influence responses of different stakeholders to global change (Nelson et al., 2006). Changes in ecosystem services also feed back to the drivers of change (e.g. altered ecosystems create new opportunities and constraints on land use) (Nelson et al., 2006).

Some effects of drivers emerge in the short-term (e.g. land use, deforestation), while others mainly in the long-term (e.g. climate change, changes in biogeochemical cycles). Long-term impacts of anthropogenic drivers of environmental change on ecosystem functioning can strongly depend on how such drivers gradually decrease biodiversity and restructure communities (Isbell et al., 2013). Current models do not account for potentially important indirect effects of habitat destruction on ecosystem services resulting from changes in biodiversity that occur within nearby remaining ecosystem fragments, even though many species could be lost from such fragments (Isbell et al., 2015).

Socio-ecological systems are characterized by causal relationships between their different components (Fischer & Christopher, 2007) Figure 4.15a and environmental problems can originate from the relationships between stakeholders, from the inefficiency of institutional arrangements in implementing regulation, from social inequality or from the inadequacy of policy actions for a given social context (Maxim et al., 2009). In addition, uncertainty is intrinsic to complex biological and social systems (Maxim et al., 2009). In the case of the Americas, reducing uncertainties through the improvement of integrated monitoring networks will enhance the ability to respond to environmental changes in the different subregions and improve the understanding of potential interactions of multiple drivers and scales and how the interactive effects of change drivers might impact (positively or negatively) ecosystem in the future.

The Figure 4.15b represents the mismatches in the spatial and temporal scales at which the relationships between anthropogenic drivers, biodiversity, and ecosystem functions and services (Isbell et al., 2017). These mismatches pose a challenge to link the cascading effects of human activities on biodiversity, ecosystems and ecosystem services. Furthermore, the scales at which knowledge is available for some of the relationships do not yet align with the scales at which policies and other decisions are often made.

Figure 4 15 A **The influence and dependence of people on biodiversity.**

People influence biodiversity directly by changing land-use, climate and biogeochemical cycles, as well as by introducing species — actions known collectively as anthropogenic drivers. At the global scale, these activities are driving the sixth mass extinction in the history of life on Earth. At the local scale, species losses decrease ecosystem functioning (for example, ecosystem productivity and resource uptake) and stability (the invariability of ecosystem productivity across a period of years). At the intermediate scales such as landscapes or regions, changes in ecosystem functioning can alter the supply of ecosystem services, including the production of wood in forests, livestock forage in grasslands and fish in aquatic ecosystems. It is important to build multiscale knowledge at the intersections of the numerous components of the system. Various system components are positioned in a gradient that spans the social (orange) to ecological (purple) ends of a socio–ecological continuum. Dashed arrows indicate other important relationships that are beyond the scope of this review. Source: Isbell *et al.* (2017).

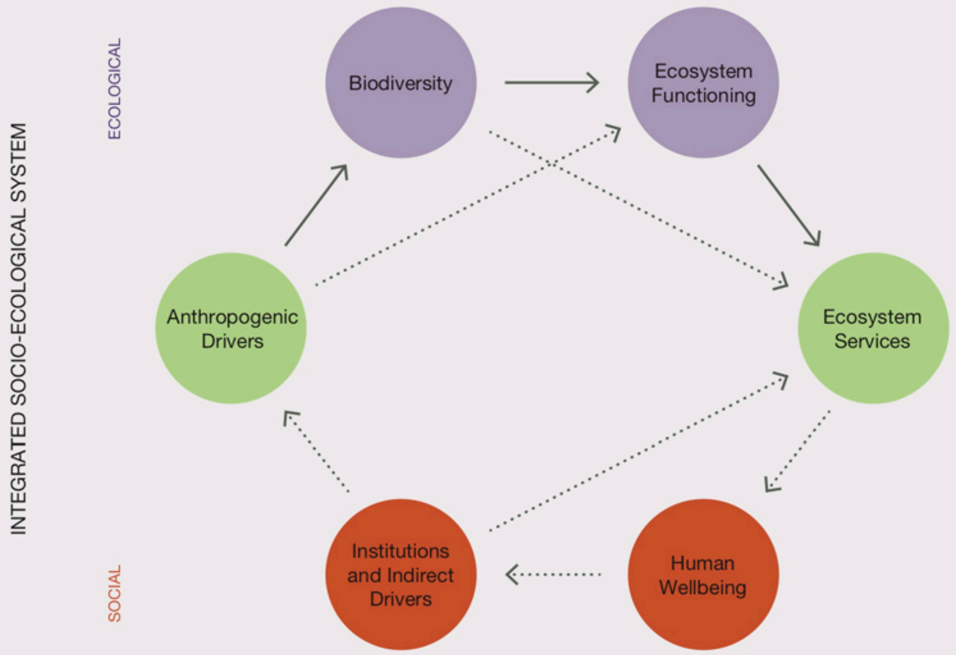
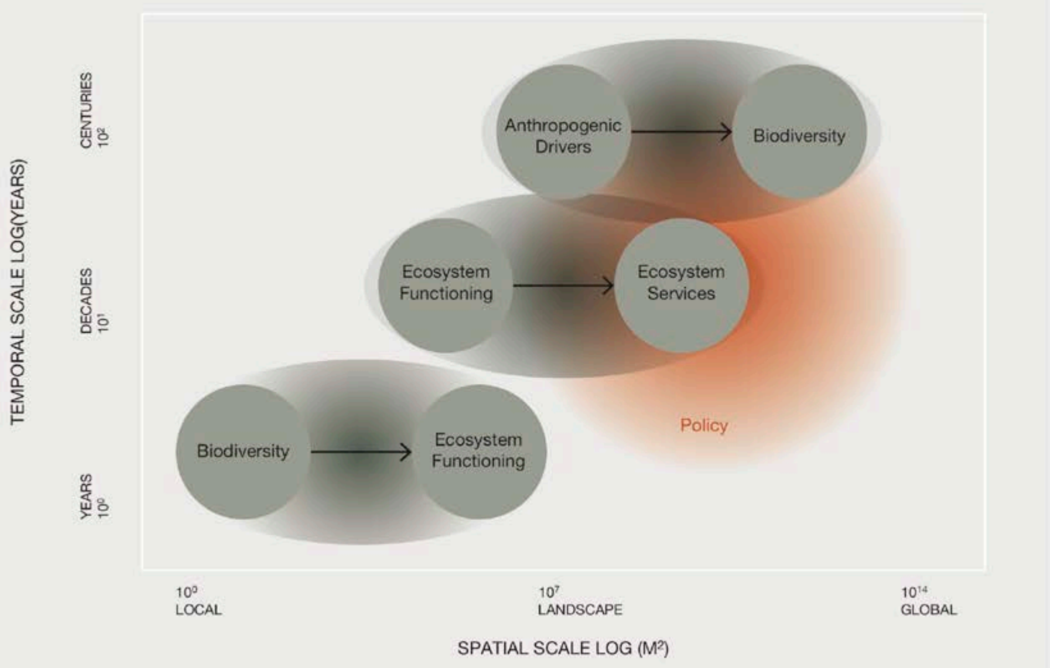


Figure 4 15 B **The influence and dependence of people on biodiversity.**

People influence biodiversity directly by changing land-use, climate and biogeochemical cycles, as well as by introducing species — actions known collectively as anthropogenic drivers. At the global scale, these activities are driving the sixth mass extinction in the history of life on Earth. At the local scale, species losses decrease ecosystem functioning (for example, ecosystem productivity and resource uptake) and stability (the invariability of ecosystem productivity across a period of years). At the intermediate scales such as landscapes or regions, changes in ecosystem functioning can alter the supply of ecosystem services, including the production of wood in forests, livestock forage in grasslands and fish in aquatic ecosystems. It is important to build multiscale knowledge at the intersections of the numerous components of the system. Various system components are positioned in a gradient that spans the social (orange) to ecological (purple) ends of a socio–ecological continuum. Dashed arrows indicate other important relationships that are beyond the scope of this review. Source: Isbell *et al.* (2017).



The Aichi 2020 targets, under the CBD, endeavor to halt the loss of biodiversity by 2020, in order to ensure that ecosystems continue to provide essential services. The present evaluation of the status and trends of the multiple drivers of change for the different units of analysis in the Americas shows that most of the Aichi targets will be not achieved without significant policy interventions. This analysis is in accordance with a study at the global scale of the many impediments for the accomplishment of the Aichi targets that indicated 15 of the Aichi targets as unlikely to be delivered; three likely to be delivered in part; and two in full (Hill et al., 2015).

Understanding and managing ecosystem-service delivery is of key importance for human wellbeing (Chapter 2). Development, poverty eradication, and biodiversity conservation are key areas of focus of the United Nations SDG. The initiative adopted in 2015 by more then 150 world leaders set targets to be achieved by 2030 as part of a new sustainable development agenda and reinforces the demand for integrated analyses of indirect and direct drivers of biodiversity and ecosystem changes. This agenda is particularly relevant to Mesoamerica and South America whose countries still show social inequality allied to economies highly dependent on the export of natural resources and agricultural commodities.

The rapidly increasing dependency on biodiversity-risk commodities, which are expanding mostly at the expense of existing natural vegetation, is currently not accompanied by comprehensive governance policies and land planning (Lemos & Agrawal, 2006). Efforts to revise this situation face a variety of challenges. The increased globalization of the world economy has catalyzed rapid growth and the complexity of international trade, leading to a disconnection and physical separation of the places of production, transformation and consumption of land-based products. This disconnectedness strongly hampers socio-environmental governance and the implementation of regulatory frameworks, beyond the intrinsic difficulties to govern sectors already in rapid transition driven by increasing global demand for food, fuel, feed and fiber Figure 4.16. As a result, natural resource use policies often come in place only when fundamental shifts in the land-use system are already underway and interventions become costly and have limited influence. Furthermore, while benefits from trade of agricultural commodities are easily measured and perceived by those in the supply chain and production countries as a whole, the associated externalities have so far been poorly understood and/or poorly translated into economic costs in future years.

The application of the knowledge of ecological and socio-ecological processes to the sustainable management of natural systems is the foundation to build resilience to future environmental change. In the different units of analysis, increasing and diverse exploitation of natural resources demands the development of different regional and national legislative initiatives aimed at protection and restoration of biodiversity and ecosystems and further adequate and sustainable management of nature (see Chapter 6). Policies and strategies could reduce the anthropogenic impacts on biodiversity by modifying the trends of drivers and underlying causes. The integration of biodiversity protection into other sectoral policies might enhance the chances for effective political action. Planning of measures to prevent and mitigate biodiversity loss, like habitat preservation, restoring degraded landscapes, maintaining or creating connectivity, avoiding overharvest, reducing fire risk and control of greenhouse gasses emissions, should consider the need to manage multiple drivers simultaneously over longer terms (Brook et al., 2008). Usually, conservation plans are developed for regions that encompass only one environmental realm (terrestrial, freshwater or marine) because of logistical, institutional and political constraints (Beger et al., 2010). However, as shown above for freshwater and wetland ecosystems, these realms often interact through processes that form, utilize and maintain interfaces or connections, which are essential for the persistence of some species and ecosystem functions. These linkages must be also considered in policy framing processes as well as the analysis of values and human behavior that induce, are affected by or respnd to the changes in environmental conditions.

Figure 4 16 Direct and indirect drivers of NCP in the Americas and their interdependencies. Source: own representation.

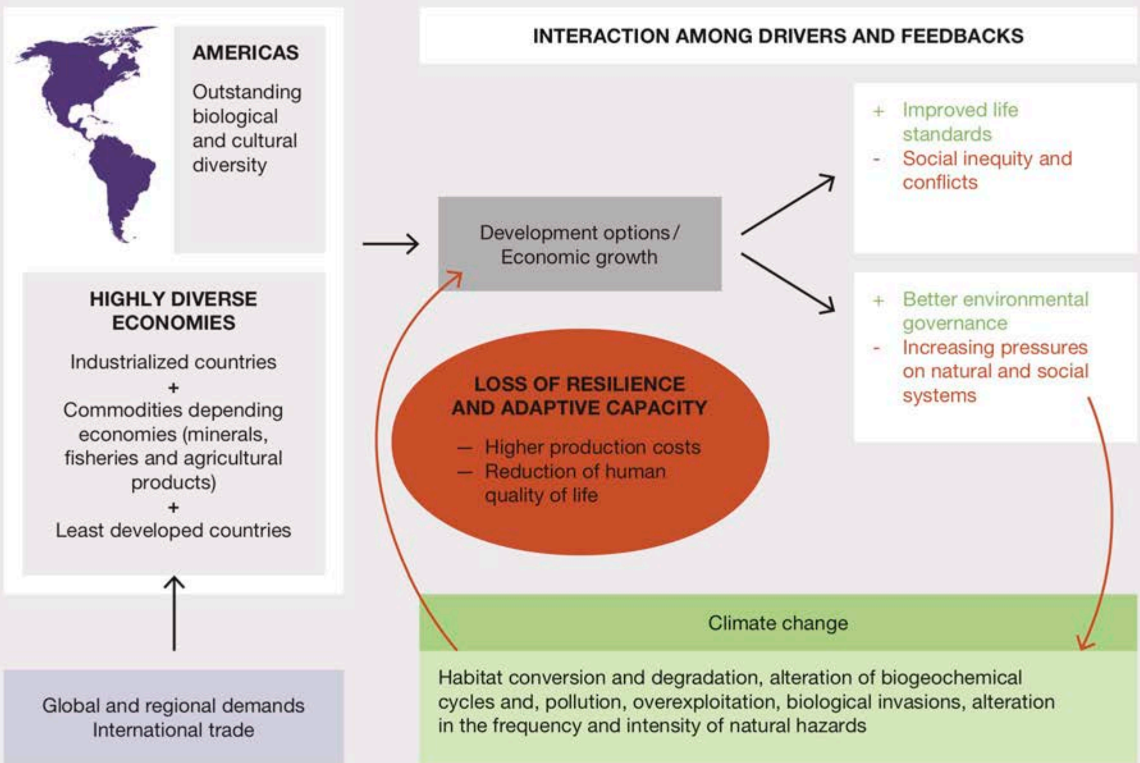


Table 4.14. Weight of the Americas in the global exports of key bodiversity-risk commodities (as percentage of global exports), 2015.

COUNTRY	Agricult. products*	Total merchandise trade*	Soy beans**	Soy oil**	Soy meal**	Meat of bovine animals; fresh or chilled**	Meat of bovine animals, frozen**	Maize**	Maize flour**	Cocoa beans**	Cocoa butter, fat and oil**	Cocoa paste**	Cotton***	Wood in the rough or roughly squared****	Wood sawn or chipped lengthwise****
Argentina	2,87	0,43	8,87	44,03	39,66	1,52	1,41	11,48	0,18	0,00	0,00	0,00	0,02	0,00	0,10
Aruba	0,01	0	0,00	0,00	0,00	0,00	0,00	0,00	0,00	0,00	0,00	0,00	0,00	0,00	0,00
Belize	0,01	0,00	0,00	0,00	0,00	0,00	0,00	0,00	0,02	0,01	0,00	0,00	0,00	0,00	0,01
Bolivia (Plurinational State of)	0,14	0,06	0,01	3,04	2,31	0,02	0,02	0,07	0,00	0,01	0,01	0,00	0,00	0,01	0,04
Brazil	6,01	1,29	41,37	13,13	22,09	3,22	17,79	19,84	17,78	0,27	3,66	1,15	0,54	0,13	1,53
Canada	3,22	2,44	3,23	1,21	0,34	5,90	1,13	0,41	0,10	0,05	0,20	0,51	0,01	5,06	0,00
Chile	0,83	0,41	0,00	0,00	0,00	0,07	0,11	0,01	0,00	0,00	0,00	0,00	0,01	0,07	3,22
Colombia	0,46	0,31	0,00	0,00	0,00	0,05	0,15	0,00	1,71	0,55	0,36	0,29	0,01	0,05	0,00
Costa Rica	0,28	0,06	0,00	0,07	0,00	0,16	0,18	0,00	0,24	0,01	0,01	0,00	0,01	0,07	0,22
Dominican Rep.	0,12	0,05	0,00	0,05	0,00	0,00	0,01	0,00	2,12	3,17	0,17	0,02	0,07	0,01	0,00
Ecuador	0,35	0,13	0,00	0,00	0,00	0,00	0,00	0,00	0,00	9,41	0,70	1,47	0,01	0,18	0,07
El Salvador	0,08	0,03	0,00	0,01	0,00	0,00	0,00	0,00	6,76	0,00	0,00	0,00	0,02	0,01	0,00
Guatemala	0,34	0,05	0,00	0,03	0,00	0,01	0,02	0,01	1,53	0,00	0,00	0,00	0,02	0,01	0,05
Honduras	0,14	0,04	na	na	na	na	na	na	na	na	na	na	na	na	na
Jamaica	0,02	0,01	0,00	0,00	0,00	0,00	0,00	0,00	0,00	0,01	0,00	0,00	0,00	0,00	0,00
Mexico	1,67	2,02	0,00	0,02	0,02	3,78	0,45	0,51	13,20	0,01	0,65	0,01	0,10	0,07	0,03
Nicaragua	0,12	0,01	0,00	0,15	0,00	0,84	1,15	0,00	0,00	0,16	0,00	0,00	0,00	0,00	0,02
Panama	0,02	0,00	0,00	0,00	0,00	0,03	0,06	0,00	0,00	0,02	0,01	0,00	0,00	0,14	0,05
Paraguay	0,42	0,05	3,48	5,49	3,72	2,57	3,30	2,25	0,00	0,00	0,00	0,00	0,00	0,01	0,03
Peru	0,30	0,22	0,00	0,00	0,00	0,00	0,00	0,01	0,02	2,36	0,94	0,31	0,03	0,00	0,10
Uruguay	0,44	0,05	2,31	0,00	0,02	1,10	4,06	0,00	0,00	0,00	0,00	0,00	0,00	6,29	0,22
USA	10,57	8,39	36,73	7,53	13,92	8,56	7,57	30,63	16,71	0,62	3,35	3,08	2,19	16,37	0,00
Venezuela (Bolivarian Republic of)	0,00	0,47	na	na	na	na	na	na	na	na	na	na	na	na	na
TOTAL VS WORLD	**28,45**	**16,54**	**96,01**	**74,78**	**82,09**	**27,81**	**37,41**	**65,23**	**60,37**	**16,65**	**10,05**	**6,86**	**3,04**	**28,49**	**5,69**

*FAOSTAT (2013), % of USA Dollars value versus world, **COMTRADE (2015), % of weight versus world, ***COMTRADE (2015), % of USA Dollars value versus world, ***COMTRADE (2015), % of m3 volume versus world

4.8 Gaps in knowledge and data

Relevant information on indirect drivers is extremely limited at environmental scales (e.g. habitats, ecosystems, biomes), which in many cases may be more relevant than institutional scales (e.g. administrative, municipalities, provinces, countries) for IPBES assessments. In addition, internationally comparable data on indirect drivers are not always available for all countries and regions of the Americas being particularly limited for small economies.

The mechanisms by which direct drivers interact are poorly understood. The mechanisms include interactions between demographic parameters, evolutionary trade-offs and synergies and threshold effects of population size and patch occupancy on population persistence. Understanding how multiple drivers of global change interact to impact biodiversity and ecosystem services requires a multiscale approach as drivers act at from global to local scales, and their interactions have emergent properties (i.e. change with the scale). The lack of appropriate research is partially due to limited data availability and analytical issues in addressing interaction effects.

In the case of the Americas, for some regions, there is still substantial uncertainty associated to spatial and temporal magnitude of the drivers (e.g. area and spatial distribution of the different land-use classes and infrastructure maps, measurements and model forecasts for climate and nitrogen deposition, distribution of invasive species). For example, studies that quantify the impacts of invasive species on biodiversity and ecosystems are still very scarse, especially outside North America. In addition, there is very little information on the effects of nitrogen deposition on tropical forests, woodlands, savannas and grasslands (Bobbink et al., 2010). Likewise, in contrast to North America, no systematic surveys exist for pollutants, including agricultural chemicals, persistent organic pollutants and mercury, in South America, the Caribbean and Mesoamerica. Another major difficult to assess the effects of pesticides on biodiversity and ecosystem services is just knowing what pesticides are used, when and how much as well as having little information on the environmental occurrence of these same pesticides. Regarding climate change, the degree to which climate change in tundra and boreal ecosystems will promote fires and droughts is not well-documented considering that these disturbances have major consequences for species productivity and dynamics in this region (Abbott et al., 2016; Pastick et al., 2017).

For some ecosystems, lack of consistent information on drivers of change is observed in all subregions of the Americas. Trends in land condition, and drivers of those trends, remain unstudied or understudied in most dryland areas across the Americas. Coastal aquatic and pelagic ocean biodiversity also remains poorly characterized throughout the Americas. Understanding how sensitive areas change in relation to regional- to global-scale processes, a mechanism to communicate the needs of people making decisions about local resources to scientists, and pathways to deliver scientific knowledge to decision makers remain prioritary needs for the region. At this time it is not possible to make a generalized statement of impacts of global changes in physical ocean dynamics and atmospheric carbon dioxide concentrations on coastal ecology. Another major unknown is the fate of plastic pollution in coastal regions of the Americas, as the amount of plastic pollution on the ocean surface is much less than the amount that is released to oceans, yet we know that many plastics can take hundreds of years to degrade (Clark et al., 2016).

A major limitation in the study and management of coastal zones around the world has been the lack of a capacity to collect, handle, and process repeated, frequent observations of aquatic and nearby wetland resources in an integrated manner to enable the detection of changes in the chemistry and in the diversity of wetland and aquatic organisms.

Regarding American mangroves, more data on consequences of nitrogen and phosphorus enrichment to nutrient cycling rates, fluxes and stocks, sediment microbial communities structure and functioning, and the resulting primary productivity in the different types of mangroves are needed, especially in underrepresented areas like South America (Reis et al., 2017). Information about oil contamination effects on sediment microbial communities and the effects of bioremediation techniques on microbial diversity in mangroves are also needed (Santos et al., 2011; Machado & Lacerda, 2004).

Improved management for overharvested species requires inventories, baselines, and monitoring knowledge of targeted species. Managers need to know population densities, sizes and trends, breeding and migration patterns, and ecological conditions they require. Understanding the threats that are causing their decline (e.g. trade markets) as well as traditional values and knowledge will assist both management and enforcement.

There are active efforts to organize partnerships and collaborations to observe biodiversity and ecosystem characteristics in the Americas. Specifically, a series of Biodiversity Observation Network efforts are being organized under the Group on Earth Observations with some of these are at the country level. Networks of regional observation systems that collaborate and share information, and that work jointly to understand biodiversity and ecosystems could provide support to existing national programs and contribute to address United Nations SDG.

4.9 Supplementary material

Box 4.17. Nutrient pollution in the Mississippi River and Gulf of Mexico

Run-off from fields used for food and fiber production, point sources of municipal waste (from human waste and manufacturing), as well asand urban run-off, can transport nutrients and sediment to rivers and streams. This can increase nutrient (phosphorus, nitrogen, and carbon) concentrations and promote algal and aquatic vegetation growth causing eutrophication.

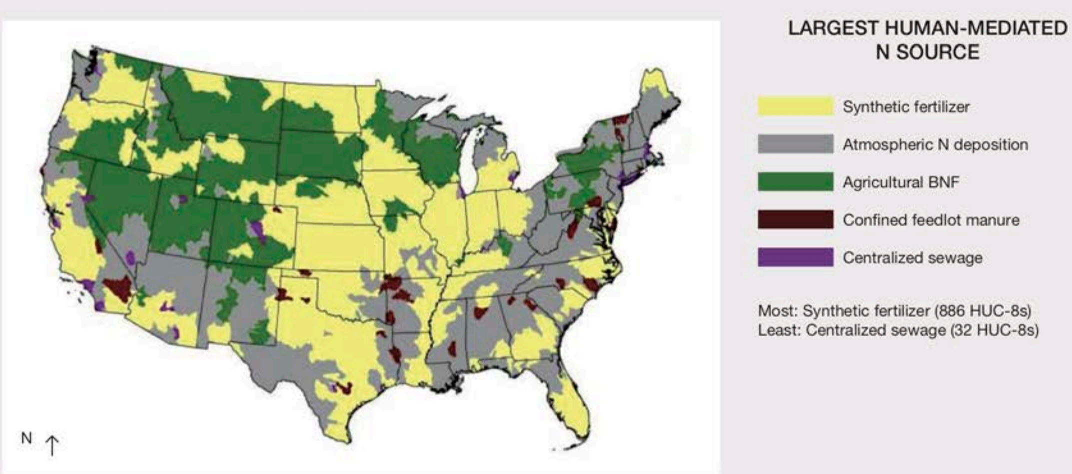
Over the last 30 years a hypoxic zone in the northern Gulf of Mexico has been measured each summer. This is an area along the Louisiana-Texas coast in which water near the bottom of the Gulf contains less than two parts per million of dissolved oxygen. Hypoxia can cause fish to leave the area disrupting fisheries and can cause stress or death to bottom dwelling organisms that can't move out of the hypoxic zone. Hypoxia is believed to be caused primarily by excess nutrients delivered from the Mississippi river in combination with seasonal stratification of Gulf waters. Excess nutrients promote algal and attendant zooplankton growth. The associated organic matter sinks to the bottom where it decomposes, consuming available oxygen. Stratification of fresh and saline waters prevents oxygen replenishment by mixing of oxygen-rich surface water with oxygen-depleted bottom water. Despite scientific concern, serious debate and billions of dollars used to ameliorate the offsite movement of nutrients in the Mississippi river basin over the past 20 years, the amount of nutrients being discharged from the Mississippi river into the Gulf of Mexico has not decreased (Sprague et al., 2011).

Shorebirds like the interior least tern and piping plover preferred habitat is sparsely vegetated sandbars along rivers or lakes and reservoir shorelines. The interior least tern was put on the Endangered Species List in the USA in 1985 and it was widely believed that river engineering threatened the species continued existence especially in the lower Mississippi river. In 2013, a Government report recommended that the interior least tern be removed from the list of plants and animals protected by the Endangered Species Act. Much of the credit for this has been given to two Federal agencies, The Fish and Wildlife Service and the Army Corps of Engineers who have specific differing responsibilities in manging the Mississippi river basin, but decided to cooperate in order to achieve objectives of flood control, navigation, and biodiversity (Nielsen, 2014).

One of the major improvements to interior least tern habitat came from a slight modification to the many engineered dikes along the lower Mississippi river which are used to focus the current into the main channel. Many of this dikes had notches built into them that allow some water through and creates backwater for fish habitat and keeps the interior least tern sand bars, isolated from shore and away from mammalian predators. Now, as Paul Hartfield, from Jackson, Mississippi, says "the interior least tern is one of the most abundant shorebirds in the lower Mississippi river" (Nielsen, 2014).

Nutrient and organic matter pollution from human sewage, urban runoff and agriculture are also a major concern in Central and South America and the Caribbean. Most municipal wastewater in South America is not treated, and rivers and estuaries draining lands with large urban areas or extensive agriculture, like the Río de la Plata, exhibit relatively high concentrations of dissolved nitrogen and organic matter (Bustamante et al., 2015; Mekonnen et al., 2015; Venturini et al., 2015). Eutrophic zones are also found in the Amazon river basin.

Figure 4 17 **Dominant sources of nitrogen to USA watershed units. Watershed units are hydrologic unit code level 8. Source: Sobota *et al.* (2013).**



Box 4.18. Organochlorine contaminant effects on bald Eagles in the Laurentian Great Lakes

Bald eagles (*Haliaeetus leucocephalus*) have been treated as bioindicator species in the recovery of the Laurentian Great Lakes from organochlorine contamination. As studies documented in early studies (Mitchell et al., 1953; Wurster et al., 1965; Wurster & Wingate, 1968) in addition to acute toxicity to songbirds, offspring of certain bird populations suffered from eggshell thinning when adults were exposed to commercial DDT. Commercial DDT is a mixture of compounds including dichlorodiphenyldichloroethylene, a much more potent toxicant towards avian populations than DDT itself. Migration surveys showed drastic declines of bald eagles from the 1940s-1960s. The species almost became extinct (Farmer et al., 2008), but populations have shown recovery since the 1970s.

The recovery of the bald eagle population in the Great Lakes was not uniform, however (Bowerman et al., 1995). Bald eagles nesting along the shores of the lakes and rivers open to spawning runs of anadromous fishes from the Great Lakes continued to exhibit impaired reproduction due to continued exposure to contaminants through consumption of contaminated fish. Total polychlorinated biphenyls, dichlorodiphenyldichloroethylene and also 2,3,7,8-tetrachlordibenzo-dioxin equivalents (TCDD-EQ; http://www.dioxinfacts.org/tri_dioxin_data/sitedata/test3/def.html) in fishes were shown to represent a significant hazard to bald eagles living along these shorelines or near the rivers. Bowerman et al. (1998) attributed the recovery of the bald eagle population along the Great Lakes to immigration of healthy individuals from interior regions. This conclusion was supported by findings that the reproduction rate of bald eagles nesting along Lake Superior's shore was significantly less than that in neighboring inland regions in Wisconsin and other inland Great Lakes sites (Dykstra et al., 1998). It was concluded that the low productivity of Lake Superior eagles was at least partly attributable to low food availability, but another factor, possibly polychlorinated biphenyls, could also have contributed to low productivity. Dykstra et al. (2001) further showed that bald eagle populations nesting on the shores of Green Bay, Lake Michigan, where concentrations of polychlorinated biphenyls are high, due to the historical presence of numerous pulp and paper mills, had reproductive rates significantly lower than those of neighboring eagles nesting inland (0.55 versus 1.1 young per occupied territory). It was concluded that organochlorine contaminants caused all or most of the depression in reproductive rates of Green Bay bald eagles.

More recently bald eagle populations have recovered. Although other contaminants, including methylmercury (Depew et al., 2013), may have sublethal or lethal effects, Dykstra et al. (2005) found that concentrations of polychlorinated biphenyls and dichlorodiphenyldichloroethylene decreased significantly in bald eagle nestling blood plasma from Lake Superior from 1989-2001. Mean concentrations were near or below threshold concentrations for reproduction impairment, and reproductive rate and contaminant concentrations were not correlated, suggesting that polychlorinated biphenyls and dichlorodiphenyldichloroethylene no longer limited Lake Superior eagle population reproduction.

Figure 4 18 Bald eagle (*Haliaeetus leucocephalus*) Photo Credit: Ron Holmes / U.S. Fish and Wildlife Service.

Box 4.19. Pollution in Greenland

Mining within Greenland is limited but related issues with pollution can occur. For example, the Black Angel mine in Maarnorilik, West Greenland, one of the richest zinc mines in the world, operated from 1973 to 1990 and restarted in 2009, has contaminated nearby waters with heavy metals especially zinc, lead, and mercury, plus others. But 30 km from the mine heavy metals are not elevated (Perner et al., 2010).

In 2004 - 2005 air samples were collected from a site in Nuuk, in Southwestern Greenland and analyzed for a suite of persistent organic pollutants. The results from the study indicate that a number of persistent organic pollutants were detected in the air in significant quantities; these included alpha and gamma hexacholorhexane, cis- and trans- chlordane, dieldrin, and degradants of DDT.

There were several studies, in two locations in Greenland that examined the long-term trends in persistent organic pollutants in biota including ringed seal, seabirds, and fish. In Greenland, there were no upwards trends in concentrations for any persistent organic pollutants, most had decreasing concentrations, although not all were statistically significant (Hung et al., 2005; Rigét et al., 2010).

In another study that examined 17 whitetail eagles found dead in Western Greenland from 1997 to 2009 all had detectable levels of persistent organic pollutants and methoxylated polybrominated diphenyl ethers in different tissues. The majority of the chemicals were found in muscle tissue and the largest portion of sum of the chemicals was polybrominated biphenyl ethers with over 50% of the totals, followed by components of DDT. Collectively the concentrations in the birds did not reach known toxic levels, but some individual birds did have levels that would be considered toxic (Jaspers et al., 2013)

In Greenland, pregnant Inuit women, women of child-bearing age and infants have high mercury and persistent organic pollutants levels in maternal blood and hair; maternal blood mercury levels exceed guidelines and are much greater compared with most Europeans; and mercury levels increase with increasing marine mammal consumption (Bjerregaard & Hansen, 2000; Dietz et al., 2013; Visnjevec et al., 2014; Weihe et al., 2002). The combined evidence suggests mercury exposure is causing subtle neurobehavioral deficits in children (Weihe et al., 2002). In the Faroe Islands, which are also in the north Atlantic, modeling suggests that mercury inputs would have to decline by ~50% to achieve safe Inuit exposure levels (Booth & Zeller, 2005), which is about the portion of the global environmental mercury burden that has man-made origins (Bergan et al., 1999). Polar bears in Greenland also have mercury levels in tissues that are high enough to be toxic. As in other Arctic biota, Greenland birds of prey have been exposed to steadily increasing levels of mercury, beginning with the industrial revolution and through the 10th century, as indicated by feather mercury levels. A few samples from the late 20th century suggest recent declines in mercury (Dietz et al., 2006).

Box 4.20. Pollution of South American mangroves

South American mangroves are threatened by human-induced alterations in the nitrogen and phosphorus cycles. Increased nitrogen availability originating from agriculture and mining activities, sewage pollution, and also from shrimp farming and direct solid waste disposal that take place in South American mangroves (Lacerda et al., 2002; Castellanos-Gallindo et al., 2014; Rodríguez-Rodríguez et al., 2016) can lead to intensification of nitrogen cycling in mangrove sediment with direct effects on ecosystem functioning and also potential indirect effects on ecosystem structure and biodiversity. As a consequence of anthropogenic nitrogen enrichment, mangroves may increase nitrous oxyde fluxes to the atmosphere, also contributing to global warming (Reis et al., 2017). Phosphorous enrichment may also extensive affect nutrient cycling in mangrove sediment by modifying physical and chemical conditions and phosphorus fractionation, and by increasing microbial activity and organic matter decomposition in sediment (Nóbrega et al., 2014). Other pollutants affecting mangroves in South America are oil spills (Lacerda & Kjerfve, 1999; Lacerda et al., 2002) and toxic metals (Machado & Lacerda, 2004). In general, consequences of oil spills to mangroves include trees defoliation and leaf deformation, mortality of seedlings and trees, bioaccumulation of toxic compounds, and reduction in faunal density, which can persist over many years after the spill (Lacerda et al., 2002). Oil spills were also reported to affect the structure and biodiversity of microbial and fungal communities in mangrove sediment (e.g. Taketani et al., 2010; Fasanella et al., 2012). Enhanced trace metal availability due to engineering works at watersheds and input of waste from urban and industrial centers and aquaculture and agriculture areas has favored trace metals trapping and storage in mangrove sediment (e.g. Machado & Lacerda, 2004; Lacerda et al., 2011; Costa et al., 2013). While the retention of such elements within mangrove sediments may contribute to the reduction of metal transfer to surrounding coastal areas, it may also cause negative effects on mangrove plants and animals, with special concerns on transfer within food chains, and transfer to man through fisheries (Machado & Lacerda, 2004).

Box 4.21. Case study: *Pterois volitans* (Linnaeus 1758) and *P. miles* (Bennett 1828) Family Scorpaenidae

The Indo-Pacific lionfish is the first nonnative marine fish to establish in the western north Atlantic and Caribbean Sea. The lionfish invasion is predicted to be the most ecologically impacting marine invasion ever recorded (Albins & Hixon, 2011). Invasive lionfish prey on a wide range of native fish species (Côté et al., 2013) due to a suite of predatory characteristics and behaviors that have no parallel in the Atlantic (Albins & Lyons, 2012; Albins & Hixon, 2013). Field experiments have demonstrated that lionfish reduced recruitment of native species in coral reef patches, including important functional groups like parrotfishes (Albins & Hixon, 2008; Green et al., 2012). The reduction in the abundance of native fishes caused by lionfish in controlled experiments was 2.5 times greater than the one caused by a similarly sized native predator (Albins, 2013), suggesting that lionfish can outcompete native predators. The first confirmed record of lionfish occurrence in the USA was a specimen taken 1985 (Morris & Akins, 2009). Whitfield et al. (2002) documented the presence and likely establishment of the Indo-Pacific lionfish *Pterois volitans* in the western Atlantic. They postulated that the source of the introduction was the marine aquarium trade. Lionfish specimens are now found along the USA east coast from Cape Hatteras, North Carolina, to Florida, and in Bermuda, The Bahamas, and the Caribbean throughout, treats including the Turks and Caicos, Haiti, Cuba, Dominican Republic, Puerto Rico, St. Croix, Belize, and Mexico (Schofield, 2009; Schofield, 2010; Betancur et al., 2011). In less than 30 years, lionfish have dramatically expanded their non-native distribution range to an area of roughly 7.3 million km^2, encompassing the eastern coast of the USA, Bermuda, the entire Caribbean region and the Gulf of Mexico (Schofield, 2010). Because of euryhaline and eurythermal features of this species, its expansion was not constrained by the Amazon-Orinoco plume (Luiz et al., 2013) and it was recently reported almost in the southeastern coast of Brazil expanding its distribution range to the Atlantic coast of South America (Ferrerira et al., 2015)

Box 4.22. Impacts of invasive alien species *Clarias* sp. on populations of freshwater fish in the biosphere Reserve Cienaga de Zapata, Cuba

Biosphere Reserve Cienaga de Zapata, is the largest wetland in the Caribbean islands and is home to high biodiversity in the presence of many local endemic. As 75% of the territory is flooded, water regime is the main ecological factor that determines the characteristics of its complex ecosystems (ACC-ICGC, 1993) .The physical, geographical and hydrological characteristics, together with the periodic floods that occur in rainy periods, and the incidence of major hurricanes, have influenced the introduction and rapid increase of two exotic and invasive species of the genus *Clarias* (*Clarias macrocephalus* and *Clarias gariepinus*), being more abundant *C. gariepinus*. This is an omnivorous species with high fertility, rapid growth and high resistance to diseases, and stress management, justifying its rapid distribution in the natural environment.

Studies for more than a decade (2003-2014) on the impact of the species on wetland biodiversity are based on the results of the analysis of stomach contents. These results showed that *C. gariepinus* feeding was mainly composed of fish in the first two years of sampling, predominantly the endemic, biajaca criolla (*Nandopsis tetracantus*) accounted for 12.5% of the diet. This species was not found in the stomach contents in the later years. Simultaneously, the analysis of the variation in the composition of catching fish companions showed that in less than two years, fish populations with some degree of endemism began to decline drastically and only introduced species maintain their populations. Importantly, from 2002, specimens of the genus *Clarias* were the most abundant in catches.

Today, populations of biajaca criolla have declined substantially in the wetland, proving to be rare in the lakes and rivers. Studies by Perez & Duarte in 1990 linked the decline in populations of biajaca criolla in Cuba with the introduction of other exotic species such as trout (*Micropterus salmoides*) and sunfish (*Lepomis macrochirus*). However, in 1979 the biajaca criolla represented 46.7% of the population of fish in Laguna del Tesoro, while 24.3% and 20.6% were trout and sunfish, respectively. It is with the arrival of specimens of the genus *Clarias* that the effects on this Cuban endemic species of freshwater fish (meat is of great commercial value), belonging the family Cichlidae became stronger (Howell Rivero & Rivas, 1940; Vales et al., 1998).

Box 4.23. More than an invasive ecosystem engineer: introduced beavers in southern Patagonia as a social-ecological system

In the 1940s and 1950s, government and private initiatives brought various exotic species to Patagonia, including Canadian beavers (*Castor canadensis*), American mink (*Neovison vison*), muskrats (*Ondatra zibethicus*), red deer (*Cervus elaphus*) and European rabbits (*Oryctolagus cuniculus*) (Ballari et al.,2016). The re-construction of this ecological landscape was largely driven by a cultural “mindscape” that valued Northern Hemisphere species over local ones, conceiving these introductions as a way to “enhance” the fauna, “develop” the region or bring “progress” to a remote area (e.g. Sucesos Argentinos) (Anonymous,1946).

Since the late 1990s, ecological research has mostly quantified the negative impacts of introduced invasive species and focused on emblematic or problematic cases like the beaver (Anderson & Valenzuela, 2014). For example, the biological invasion by beavers has been shown to be a significant transformation of sub-Antarctic forests in the Holocene. As an invasive ecosystem engineer, the beaver creates novel ecosystems conformed by meadows and ponds that reorganize biotic communities and facilitate the spread of other exotic flora and fauna, but they also provide habitat for native waterfowl and fish (Anderson et al., 2014). However, unlike the northern hemisphere, southern Patagonian forests in particular are not resilient to beaver impacts, and therefore, they require active restoration measures to ameliorate beaver impacts (Wallem et al., 2010). This ecological information motivated Argentine and Chilean decision-makers to agree to eradicate beavers and restore degraded ecosystems. However, it quickly became apparent that achieving these goals required understanding not only ecological dimensions, but also social aspects of this system. Although global images of Patagonia tend to project it as an unsullied wilderness, but it has a long history of human habitation and a modern social context that is quite complex (Moss, 2008). In the case of beavers, an eradication program must recognize that the Tierra del Fuego Archipelago is one biogeographic unit, but it is administered by two nations with different political-administrative systems. Furthermore, different social groups within each country understand their relationship with beavers differently. For example, while environmental managers in southern Patagonia rank invasive species as a primary threat to ecosystems, the 98% of residents who live in cities do not perceive them as a priority problem (Zagarola et al., 2014). Indeed, the novel social context of beavers includes the fact that they have become a symbol for various tourism enterprises and companies, particularly in Argentina. This social system includes not only two nation-States, but diverse stakeholders and social groups that have multi-relationships and perspectives with this multi-natural ecosystem (Santo et al., 2015). Incorporating this complexity of human and environmental factors means reconceiving biological invasions and restoration ecology as social-ecological systems for both research and management, but achieving this recognition has literally taken decades. By recognizing the social-ecological dimensions of invasive exotic species, not just their "biological invasion", ecologists would be better positioned to effectively and efficiently address these and other problems in association with not only other academic disciplines, but other social actors that are part of the study and management of environmental issues.

Box 4.24. Case study: *Limnoperna fortunei* (Dunker, 1857)

This mussel species, commonly known as the golden mussel, is native to the freshwater systems southeast China. Because of the ecological effects caused in aquatic ecosystems and expenses incurred in industrial infrastructure concerned is considered as aquatic invasive species and environmental issues at regional level (Darrigran, 2002). It was accidentally introduced to the region of the Río de la Plata basin in 1991 through ballast water and first reported on the coast of Río de la Plata, Buenos Aires (Pastorino et al., 1993, Darrigran & Pastorino, 1995). Currently, it has a rapid ascent up the Río de la Plata basin (feed rates of 250 km per year), invading major rivers (Río de la Plata, Uruguay, Parana, Paraguay, Tiete) and smaller water systems in basins Guaíba, Tramandaí (south east Brazil), Laguna de los Patos-Mirim (Brazil-Uruguay), Mar Chiquita (Argentina-central) or Laguna del Sauce (east coast Uruguay) (de Oliveria et al., 2015). It is currently in aquatic environments from five countries in South America: Argentina, Brazil, Bolivia, Paraguay and Uruguay, identified as the main vector of invasion commercial navigation on the waterway of the Río de la Plata basin (Karatayev et al., 2006). Since its arrival to the region, it was found associated with a variety of natural and artificial substrates consolidated, increasing its population abundances, causing changes in the benthic communities and in the eating habits of native fish. It generates further problems macrofouling (settlement and colonization of organisms greater than 50 micrometre on artificial substrates) in hydraulic systems of companies and industries that use different branches water resources in their production cycles (Boltovskoy & Correa, 2014). Among the effects caused are clogging of filters, disablement of hydraulic sensors, damages to pumps or decreased uptake diameter line pipe for cooling water, irrigation, or water purification. These effects cause overhead in major water purification water plants, nuclear, hydroelectric plants, refineries, steel mills and agro-industrial plants (aquaculture, forestry, food), due to maintenance, structural modifications, as well as management plans and population control (Brugnoli et al., 2006; Boltovskoy & Correa, 2014; Boltovskoy et al., 2015).

Box 4.25. Case study: *Rapana venosa* (Valenciennes, 1846)

The snail rapana is native to the Sea of Japan, Yellow Sea, Bohai Sea and the Sea of China to Taiwan (Mann et al., 2004). In 1947, it was described for the first time outside of its original range in the Black Sea and then subsequently reported in the Azov, Aegean, Adriatic Seas and North America (Pastorino et al., 2000, Mann et al., 2004, Kerckhof et al., 2006). It is a predator of molluscs subtidal, usually feeding on bivalves of economic interest such as oysters, mussels and clams (Harding & Mann, 1999; Savini & Occhipinti-Ambrogi, 2006; Giberto et al., 2011; Lanfranconi et al., 2013).

It was first recorded in South America in 1999 in the Río de la Plata, Argentinian coast (Bay Samborombón) (Pastorino et al., 2000). A decade after its first records outside Samborombón Bay, the species expanded its distribution to all muddy bottoms of the subtidal mixohaline zone of the Río de la Plata (Giberto et al., 2006). For the Uruguayan coast of the Río de la Plata, Scarabino et al. (1999) reported on the coast of Maldonado; meanwhile, Carranza et al. (2007) describe its distribution in the outer area of the Río de la Plata. Currently, it presents its limit of this distribution in the Bay of Maldonado-Punta del Este (Lanfranconi et al., 2009; Carranza et al., 2010).

Perception of local communities: conducting a study with a multidisciplinary approach involving biologists, sociologists and consultation of fisherfolk (mussel) in the south east of Uruguay coast, allowed to highlight the importance of considering local knowledge with stakeholders involved daily with the impact of invasive species on fishery resources (Brugnoli et al., 2014). The "empirical" knowledge, largely consolidates existing scientific knowledge concerning *R. venosa* and, in certain cases, brings new questions for future research. Both approaches (scientific-community local) agree on the dates of the first observations of the snail to the area as well as observation of mucous trail left by its movement. This empirical knowledge as well as information collected in the field by local people, is sometimes prescinded by the academy. However, it could play an important role in monitoring programs that include early warning, monitoring of abundance and distribution, as well as the identification of direct or indirect effects on the native fauna caused by invading organisms like *R. venosa*

4.10 References

Abbott, B. W., Jones, J., Schuur, E., Chapin III, F. S., Bowden, W., Bret-Harte, M., Epstein, H., Flannigan, M., Harms, T., Hollingworth, T., Mack, M., McGuire, A. D., Natali, S., Rocha, A., Tank, S., Turetsky, M., Vonk, J. E., Wickland, K. P., Aiken, G. R., Alexander, H., Amon, R. M. W., Welker, J., & Zimov, S. (2016). Biomass offsets little or none of permafrost carbon release from soils, streams, and wildfire: an expert assessment. Environmental Research Letters, 11(3), 34014. https://doi.org/10.1088/1748-9326/11/3/034014

Abreu, R. C. R. de, & Durigan, G. (2011). Changes in the plant community of a Brazilian grassland savannah after 22 years of invasion by Pinus elliottii Engelm. *Plant Ecology & Diversity*, 4(2–3), 269–278. https://doi.org/10.1080/17550874.2011.594101.

ACC-ICGC. (1993). Estudio Geográfico Integral. Ciénaga de Zapata. Publicaciones del Servicio de Información y Traducciones, La Habana, Cuba.

Achard, F., Eva, H. D., Stibig, H. J., Mayaux, P., Gallego, J., Richards, T., & Malingreau, J. P. (2002). Determination of deforestation rates of the world's humid tropical forests. *Science,* 297(5583), (pp. 999-1002).

Achard, F., Eva, H., Glinni, A., Mayaux, P., Richards, T., & Stibig, H. J. (1998). Identification of deforestation hot spot areas in the humid tropics. *Trees Publication Series B*, 4, (pp.1–81).

Adkins, Collette. (2016). The US Endangered Species Act: a powerful tool to protect biodiversity (if we use it). *Biodiversity*, 17.3 (2016). (pp. 101-103).

Affonso, A G., Barbosa, C., & Novo, E. M. L. M. (2011). Water quality changes in floodplain lakes due to the Amazon River flood pulse: Lago Grande de Curuaí (Pará). *Brazilian journal of biology* (Revista brasleira de biologia), 71:601–10. <http://www.ncbi.nlm.nih.gov/pubmed/21881783>.

Agard, J., Cropper, A., Aquing, P., Attzs, M., Arias, F., Beltran, J., Bennett, E., Carnegie, R., Clauzel, S., Corredor, J., & Creary, M. (2007). Caribbean Sea ecosystem assessment (CARSEA). *Caribbean Mar Stud*, 8, (pp.1-85).

Agencia EFE. (1998). Anuario Iberoamericano. Agencia EFE / ANUARIO. Madrid . 503 p.

Agnew, D. J., Gutiérrez, N. L., Stern-Pirlot, A., & Hoggarth, D. D. (2014). The MSC experience: Developing an operational certification standard and a market incentive to improve fishery sustainability. *ICES Journal of Marine Science*, 71(2), (pp. 216–225).

Seebens, H., Blackburn, T. M., Dyer, E. E., Genovesi, P., Hulme, P. E., Jeschke, J. M., Pagad, S., Pyšek, P., Winter, M., Arianoutsou, M., Bacher, S., Blasius, B., Brundu, G., Capinha, C., Celesti-Grapow, L., Dawson, W., Dullinger, S., Fuentes, N., Jäger, H., Kartesz, J., Kenis, M., Kreft, H., Kühn, I., Lenzner, B., Liebhold, A., Mosena, A., Moser, D., Nishino, M., Pearman, D., Pergl, J., Rabitsch, W., Rojas-Sandoval, J., Roques, A., Rorke, S., Rossinelli, S., Roy, H. E., Scalera, R., Schindler, S., Štajerová, K., Tokarska-Guzik, B., van Kleunen, M., Walker, K., Weigelt, P., Yamanaka, T., & Essl, F. (2017). No saturation in the accumulation of alien species worldwide. *Nature Communications,* 8, 14435. https://doi.org/10.1038/ncomms14435

Ahlström, A., Raupach, M. R., Schurgers, G., Smith, B., Arneth, A., Jung, M., Reichstein, M., Canadell, J. G., Friedlingstein, P., Jain, A. K., & Kato, E. (2015). The dominant role of semi-arid ecosystems in the trend and variability of the land CO2 sink. *Science,* 348, (pp. 895–899).

Aide, T. M., Clark, M. L., Grau, H. R., Lopez-Carr, D., Levy, M. A., Redo, D., Bonilla-Moheno, M., Riner, G., Andrade-Nuñez, M. J., & Muñiz, M. (2013). Deforestation and reforestation of Latin America and the Caribbean (2001 – 2010). *Biotropica*, 45(2), (pp. 262–271). https://doi.org/10.1111/j.1744-7429.2012.00908.x

Ajanovic, A. (2011). Biofuels versus food production: does biofuels production increase food prices? *Energy*, 36(4), (pp. 2070-2076).

Filip, O., Janda, K., Kristoufek, L., & Zilberman, D. (2017). Food versus fuel: An updated and expanded evidence. *Energy Economics*.

Albert, D. (2000). Borne of the wind: an introduction to the ecology of Michigan sand dunes. Michigan Natural Features Inventory, East Lansing, Michigan.

Albins, M. A. & Hixon, M. A. (2013). Worst-case scenario: potential long-term effects of invasive predatory lionfish (*Pterois volitans*) on Atlantic and Caribbean coral-reef communities. *Environmental Biology of Fishes,* 96(10-11), (pp.1151-1157) DOI: 10.1007/s10641-011-9795-1.

Albins, M. A. (2013). Effects of invasive Pacific red lionfish *Pterois volitans* versus a native predator on Bahamian coral-reef fish communities. *Biol Invasions,* 15, (pp. 29−43).

Albins, M. A. & Hixon, M. A. (2008). Invasive Indo-Pacific lionfish *Pterois volitans* reduce recruitment of Atlantic coral-reef fishes. *Marine Ecology Progress Series,* 367, (pp. 233–238).

Albins, M. A. & Hixon, M. A. (2011). Worst case scenario: Potential long-term effects of invasive predatory lionfish (*Pterois volitans*) on Atlantic and Caribbean coralreef communities. *Environmental Biology of Fishes*, Volume 96, Issue 10–11, (pp. 1151–1157)
DOI 10.1007/s10641-011- 9795-1.

Albins, M. A. & Lyons, P. J. (2012). Invasive red lionfish *Pterois volitans* blow directed jets of water at prey fish. *Marine Ecology Progress Series*, 448, (pp. 1−5).

Albright, R., & Langdon, C. (2011). Ocean acidification impacts multiple early life history processes of the Caribbean coral *Porites astreoides*. *Global Change Biology,* 17, (pp. 2478–2487).

Alcamo, J., Vuuren, D. Van, Cramer, W., Alder, J., Bennett, E., Carpenter, S., Christensen, V., Foley, J., Masui, T., Morita, T., Neill, B. O., Peterson, G., Ringler, C., Schulze, K., Bouwman, L., Eickhout, B., Floerke, M., Lal, R., Takahashi, K., Editors, R., Tan, B., Hammond, A., & Field, C. (2005). Changes in ecosystem services and their drivers across the scenarios. Ecosystems and Human Well-Being: Scenarios, Volume 2 (2), (pp. 297–373).

Alencar, A. A., Brando, P. M., Asner, G. P., & Putz, F. E. (2015). Landscape fragmentation, severe drought, and the new Amazon forest fire regime. *Ecological applications,* 25(6), (pp. 1493-1505).

Alencar, A., Nepstad, D., & Vera Diaz, M. C. (2006). Forest understory fire in the Brazilian Amazon in ENSO and non-ENSO years: area burned and committed carbon emissions. *Earth Interactions* 10, (pp.1–17).

Alexander, C., Bynum, N., Johnson, E., King, U., Mustonen, T., Neofotis, P., Oettlé, N., Rosenzweig, C., Sakakibara, C., Shadrin, V., Vicarelli, M., Waterhouse, J., & Weeks, B. (2011). Linking Indigenous and scientific knowledge of climate change. *BioScience,* 61(6), 477–484. https://doi.org/10.1525/bio.2011.61.6.10

Alexander, J. M., Lembrechts, J. J., Cavieres, L. A., Daehler, C., Haider, S., Kueffer, C., Liu, G., McDougall, K., Milbau, A., Pauchard, A., Rew, L. J., & Seipel, T. (2016). Plant invasions into mountains and alpine ecosystems: current status and future challenges. *Alpine Botany*, 126(2), (pp. 89–103). https://doi.org/10.1007/s00035-016-0172-8

Alho, C., Reis, R., Aquino, P., 2015. Amazonian freshwater habitats experiencing environmental and socioeconomic threats affecting subsistence fisheries. *Ambio*, 44, (pp. 412–425).

Alig, R. J., Kline, J. D., & Lichtenstein, M. (2004). Urbanization on the US landscape: looking ahead in the 21st century. *Landscape and urban planning,* 69(2), (pp. 219-234).

Alkemade, R., van Oorschot, M., Miles, L., Nellemann, C., Bakkenes, M., & Ten Brink, B. (2009). GLOBIO3: a framework to investigate options for reducing global terrestrial biodiversity loss. *Ecosystems*, 12(3), (pp. 374-390).

Allan, J. D., McIntyre, P. B., Smith, S. D. P., Halpern, B. S., Boyer, G. L., Buchsbaum, A., Burton, G. A., Campbell, L. M., Chadderton, W. L., Ciborowski, J. J. H., Doran, P. J., Eder, T., Infante, D. M., Johnson, L. B., Joseph, C. A., Marino, A. L., Prusevich, A., Read, J. G., Rose, J. B., Rutherford, E. S., Sowa, S. P., & Steinman, A. D. (2013). Joint analysis of stressors and ecosystem services to enhance restoration effectiveness. *Proceedings of the National Academy of Sciences of the United States of America*, 110(1), (pp. 372–377). https://doi.org/10.1073/pnas.1213841110

Allan, J. D., Abell, R. Hogan, Z., Revenga, C., Taylor, B. W., Welcomme, R. L., & Winemiller, K. (2005). Overfishing of Inland Waters. *BioScience* 55, (pp. 1041–1051).

Almeida-Neto, M., Prado, P. I., & Lewinsohn, T. M. (2011). Phytophagous insect fauna tracks host plant responses to exotic grass invasion. *Oecologia*, 165(4), (pp. 1051–1062). https://doi.org/10.1007/s00442-010-1783-1

Alongi, D. M. (2008). Mangrove forests: resilience, protection from tsunamis, and responses to global climate change. *Estuarine, Coastal and Shelf Science,* 76, (pp. 1-13).

Alongi, D. M. 2015. The impact of climate change on mangrove forests. *Current Climate Change Reports,* 1, (pp. 30-39).

Altieri, M.A., & Funes-Monzote, F. R. (2012). The paradox of Cuban agriculture. *Monthly Review*, 63(8), (p.23).

Altieri, M.A., Funes-Monzote, F.R., & Petersen, P. (2012). Agroecologically efficient agricultural systems for smallholder farmers: contributions to food sovereignty. *Agronomy for Sustainable Development*, 32(1), (pp.1-13).

Álvarez-Berríos, N. L., Redo, D. J., Aide, T. M., Clark, M. L., & Grau, R. (2013). Land Change in the Greater Antilles between 2001 and 2010. *Land*, 2(2), (pp. 81-107).

Alvarez-Filip, L., Dulvy, N. K., Gill, J. A., Côté, I. M., & Watkinson, A. R. (2009). Flattening of Caribbean coral reefs: region-wide declines in architectural complexity. *Proceedings. Biological sciences / The Royal Society* 276 (1669), (pp. 3019–25). doi: 10.1098/rspb.2009.0339

Anderson, C. B., Griffith, C. R., Rosemond, A. D., Rozzi, R., & Dollenz, O. (2006). The effects of invasive North American beavers on riparian plant communities in Cape Horn, Chile: Do exotic beavers engineer differently in sub-Antarctic ecosystems? *Biological Conservation*, 128(4), (pp. 467–474). https://doi.org/10.1016/j.biocon.2005.10.011

Anderson, C.B. & Valenzuela, A.E.J. (2014). Do what I say, not what I do. Are we linking research and decision-making about invasive species in Patagonia? *Ecología Austral,* 24, (pp. 193-202).

Anderson, C.B., Lencinas, M.V., Valenzuela, A.E.J., Simononok, M.P., Wallem, P.K., & Martinez P., G. (2014). Ecosystem engineering by an invasive species, the beaver, increases landscape-level ecosystem function but does not affect biodiversity in Tierra del Fuego's freshwater systems. *Diversity and Distributions*, 20, (pp. 214–222).

Anderson, E. P. (2013). Hydropower development and ecosystem services in Central America. Inter-American Development Bank. Technical note No. IDB-TN-513

Anderson, O. R. J., Small, C. J., Croxall, J. P., Dunn, E. K., Sullivan, B. J., Yates, O., & Black, A. (2011). Global seabird bycatch in longline fisheries. *Endangered Species Research,* 14 (pp.91–106).

Anderson, O. J., & Piatt, J. F. (1999). Community reorganization in the Gulf of Alaska following ocean climate regime shift. *Marine Ecology Progress Series*, vol. 189, (pp. 117-123).

Anderson-Teixeira, K. J., Snyder, P. K., Twine, T. E., Cuadra, S. V., Costa, M. H., & Delucia, E. H. (2012). Climate-regulation services of natural and agricultural ecoregions of the Americas. *Nature Climate Change* 2, (pp. 177–181).

Angel, J. R., & Huff, F. A. (1997). Changes in heavy rainfall in midwestern U.S. *Journal of Water Resources Planning and Management* - ASCE 123 (4), (pp.246–249). https://doi.org/10.1061/(ASCE)0733-9496(1997)123:4(246)

Angel, J. R., & Kunkel, K. E. (2010). The response of Great Lakes water levels to future climate scenarios with an emphasis on Lake Michigan-Huron. *Journal of Great Lakes Research* 36, (pp. 51–58). http://dx.doi.org/10.1016/j.jglr.2009.09.006.

Angelelli, P., & Saffache, P. (2013). Some remarks on mangroves in the Lesser Antilles. *Revista de Gestão Costeira Integrada (Journal of Integrated Coastal Zone Management)*, 13(4).

Anonymous. (1946). Viaje al Sur. *Sucesos Argentinos,* 432. Government Newsreel. Buenos Aires.

Anthony, K. R. N., Maynard, J. A. , Diaz-Pulido, G., Mumby, P. J., Marshall, P. A., Cao, L., & Hoegh-Guldberg, O. (2011). Ocean acidification and warming will lower coral reef resilience. *Global Change Biology,* 17, (pp. 1798–1808).

Appeldoorn, R. S., Gonzalez, E. C., Glazer, R., & Prada, M. (2011). Applying EBM to queen conch fisheries in the Caribbean. (pp.177–186). In: Fanning, L., Maho, R., & McConney, P. (editors). Towards Marine Ecosystem-based Management in the Caribbean.

Apte, J. S., Marshall, J. D., Cohen, A. J., & Brauer, M., (2015). Addressing global mortality from ambient PM2. 5. *Environmental science & technology*, 49(13), (pp.8057-8066).

Archer, S., Schimel, D. S., & Holland, E. A. (1995). Mechanisms of shrubland expansion: land use, climate or CO2? *Climatic Change*, 29(1), (pp. 91–99). https://doi.org/10.1007/BF01091640

Armendáriz-Villegas, E. J., Covarrubias-García, M. D. L. A., Troyo-Diéguez, E. , Lagunes, E., Arreola-Lizárraga, A., Nieto-Garibay, A., Beltrán-Morales, L.F., & Ortega-Rubio, A. (2015). Metal mining and natural protected areas in Mexico: Geographic overlaps and environmental implications. *Environmental Science and Policy*, 48, (pp. 9-19).

Armenteras-Pascual, D., Retana-Alumbreros, J., Molowny-Horas, R., Roman-Cuesta, R. M., Gonzalez-Alonso, F., & Morales-Rivas, M. (2011). Characterising fire spatial pattern interactions with climate and vegetation in Colombia. *Agricultural and Forest Meteorology*, 151(3), (pp. 279-289).

Armitage, D., De Loë, R., & Plummer, R. (2012). Environmental governance and its implications for conservation practice. *Conservation Letters*, 5(4), (pp. 245–255). http://doi.org/10.1111/j.1755-263X.2012.00238.x

Aronson, M. F. J., La Sorte, F. A., Nilon, C. H., Katti, M., Goddard, M. A., Lepczyk, C. A., Warren, P. S., Williams, N. S. G., Cilliers, S., Clarkson, B., Dobbs, C., Dolan, R., Hedblom, M., Klotz, S., Kooijmans, J. L., Kuhn, I., MacGregor-Fors, I., McDonnell, M., Mortberg, U., Pysek, P., Siebert, S., Sushinsky, J., Werner, P., & Winter, M. (2014). A global analysis of the impacts of urbanization on bird and plant diversity reveals key anthropogenic drivers. *Proceedings of the Royal Society B: Biological Sciences*, 281(20133330). https://doi.org/10.1098/rspb.2013.3330

Arriaga, L. (2000). Types and causes of tree mortality in a tropical montane cloud forest of Tamaulipas, Mexico. *Journal of Tropical Ecology*, 16(5), (PP. 623-636).

Arriaga, L., Castellanos, A. E., Moreno, E., & Alarcon, J. (2004). Potential ecological distribution of alien invasive species and risk assessment: a case study of buffel grass in arid regions of Mexico. *Conservation Biology*, 18(6), (PP. 1504–1514). https://doi.org/DOI 10.1111/j.1523-1739.2004.00166.x

Arsel, M., Hogenboom, B., & Pellegrini, L. (2016). The extractive imperative and the boom in environmental conflicts at the end of the progressive cycle in Latin America. *Extractive Industries and Society*, 3(4), 877–879.

Astor, Y. M., Lorenzoni, L .B., Thunell, R., Varela, R., Muller-Karger, F., Troccoli, L., Taylor, G. T., Scranton, M. I., Tappa, E., & Rueda, D. (2013). Interannual variability in sea surface temperature and fCO2 changes in the Cariaco Basin. *Deep-Sea Research II, special issue of Ocean Biogeochemistry Time Series Research*. Doi 10.1016/j.dsr2.2013.01.002.

Avcı, D., & Fernández-Salvador, C. (2016). Territorial dynamics and local resistance: Two mining conflicts in Ecuador compared. *Extractive Industries and Society*, 3(4), (PP. 912–921). http://doi.org/10.1016/j.exis.2016.10.007

Ávila, S., Muñoz, C., Jaramillo, L., Martínez, A. (2005). Un análisis del subsidio a la tarifa 09 Gaceta Ecológica, 75, abril-junio, Secretaría de Medio Ambiente y Recursos Naturales Distrito Federal, México. (pp. 65-76)

Baker, D. B., Richards, R. P., Loftus, T. T., & Kramer. J. W. (2004). A new flashiness index: Characteristics and applications to midwestern rivers and streams. *Journal of the American Water Resources Association*, 40, (pp. 503–522).

Baldi, G., & Paruelo, J. (2008). Land-use and land cover dynamics in South American temperate grasslands. *Ecology and Society*, 13(2).

Ballari, S.A., Anderson , C. B. & Valenzuela, A. E. J. (2016). Invasive exotic mammal research and management in the Southern Cone. *Mammal Review*, 46, (pp. 229-240).

Balvanera, P., Cotler, H., Aburto, O., Aguilar, A., Aguilera, M., Aluja, M., Andrade, A., Arroyo, I., Ashworth, L., Astier, L., Ávila, P., Bitrán, D., Camargo, T., Campo, J., Cárdenas, B., Casas, A., Díaz-Fleischer, F., Etchevers, J., Ghillardi, A., González-Padilla, E., • Guevara, A., Lazos, E., López, C., López, R., Martínez, J., Masera, O., Mazari, M., Nadal, A., Pérez-Salicrup, D., Pérez-Gil, R., Quesada, M., Ramos-Elorduy, J., Robles, A., Rodríguez, H., Rull, J. Suzán, G., Vergara, C., Xolalpa, S., Zambrano, L., & Zarco, A. (2009). Estado y tendencias de los servicios ecosistémicos. *Capital natural de México*, 2, (pp. 185-245). Retrieved from http://www.biodiversidad.gob.mx/pais/pdf/CapNatMex/Vol%20II/II04_EdoTendenciasServiciosEcosistemicos.pdf

Barbarán, F. R., Rojas, L., & Arias, H. M. (2015). Sostenibilidad institucional y social de la expansión de la frontera agropecuaria. *Revibec: revista de la Red Iberoamericana de Economia Ecológica*, 24, (pp. 0021-37).

Barbaran, F. (2015). Biodiversity values and payment for ecosystem services in Argentina: Who pays? IUCN Commission on Environmental, Economic and Social Policy (CEESP). *CEESP Newsletter*, 1515. http://iucn.org/union/commissions/ceesp/?21652

Barbour, A. B., Montgomery, M. L., Adamson, A. A., Díaz-Ferguson, E., & Silliman, B. R. (2010). Mangrove use by the invasive lionfish Pterois volitans. *Marine Ecology Progress Series*, 401, (pp. 291–294). https://doi.org/10.3354/meps08373

Barkley, A. N. (2015). Implications of developing deep-sea Arctic fisheries for Greenland Halibut (*Reinhardtius hippoglossoides*): Inshore stock connectivity and capture induced stress of ecologically important fish species. University of Windsor. *Electronic Theses and Dissertations.* Paper 5429.

Barona, E., Ramankutty, N., Hyman, G., & Coomes, O. T. (2010). The role of pasture and soybean in deforestation of the Brazilian Amazon. *Environmental Research Letters*, 5(2), 024002.

Barral, M. P., Rey Benayas, J. M., Meli, P. & Maceira, N. O. (2015). Quantifying the impacts of ecological restoration on biodiversity and ecosystem services in agroecosystems: A global meta-analysis. *Agriculture, Ecosystems and Environment,* 202, (pp.223–231).

Barros, V. R., Boninsegna, J. A. Camilloni, I. A., Chidiak, M., Magrín, G. O., & Rusticucci, M. (2015). Climate change in Argentina: Trends, projections, impacts and adaptation. Wiley Interdisciplinary Reviews. *Climate Change,* 6, (pp.151–169).

Bartrons, M., Catalan, J. & Penuelas, J. (2016). Spatial and temporal trends of organic pollutants in vegetation from remote and rural areas. *Scientific reports*, 6.

Basili, G. D., & Temple, S. A. (1999). Dickcissels and crop damage in Venezuela: defining the problem with ecological models. *Ecological applications*, 9(2), (pp. 732-739).

Bates, N., Astor, Y., Church, M., Currie, K., Dore, J., Gonaález-Dávila, M., Lorenzoni, L., Muller-Karger, F., Olafsson, J., & Santa-Casiano, M. (2014). A time-series view of changing ocean chemistry due to ocean uptake of anthropogenic CO2 and ocean acidification. *Oceanography*, 27(1), (pp. 126–141). https://doi.org/10.5670/oceanog.2014.16

Battaglin, W. A., Meyer, M. T., Kuivila, K. M. & Dietze, J. E. (2014). Glyphosate and its degradation product AMPA occur frequently and widely in US soils, surface water, groundwater, and precipitation. *JAWRA Journal of the American Water Resources Association*, 50(2), (pp.275-290).

Batzer, D. P., & Baldwin, A. H. (2012). Wetland habitats of North America: ecology and conservation concerns. In: Wetland Habitats of North America: Ecology and Conservation Issues, 405. https://doi.org/10.1899/32.1.BR.359.1

Batzer, D. P. & Baldwin, A. H. (2012). Wetlands habitats of North America: Ecology and conservation concerns. Univ. California Press, ISBN- 978-0520-27164-7. (pp. 387).

Baud, M., Castro, F. de, & Hogenboom, B. (2011). Environmental governance in Latin America: Towards an integrative research agenda. *European Review of Latin American and Caribbean Studies*, 90, (pp. 79–88). http://doi.org/10.18352/erlacs.9749

Bauhardt, C. (2014). Solutions to the crisis? The green new deal, degrowth, and the solidarity economy: Alternatives to the capitalist growth economy from an ecofeminist economics perspective. *Ecological Economics*, 102, (pp. 60–68). http://doi.org/10.1016/j.ecolecon.2014.03.015

Beck, M. W., Brumbaugh, R. D., Airoldi, L., Carranza, A., Coen, L. D., Crawford, C., Defeo, O., Edgar, G. J., Hancock, B., Kay, M. C., Lenihan, H. S., Luckenbach, M. W., Toropova, C. L., Zhang, G. & Guo, X. (2011). Oyster reefs at risk and recommendations for conservation, restoration, and management. *BioScience* 61, (pp. 107–116).

Bednaršek, N., Harvey, C. J., Kaplan, I. C., Feely, R. A., & Možina, J. (2016). Pteropods on the edge: Cumulative effects of ocean acidification, warming, and deoxygenation. *Progress in Oceanography*, 145, (pp. 1–24). https://doi.org/10.1016/j.pocean.2016.04.002

Beger, M., Grantham, H. S., Pressey, R. L., Wilson, K. A., Peterson, E. L., Dorfman, D., Mumby, P. J., Lourival, R., Brumbaugh, D. R., & Possingham, H. P. (2010). Conservation planning for connectivity across marine, freshwater, and terrestrial realms. *Biological Conservation,* 143(3), (pp. 565–575) . https://doi.org/10.1016/j.biocon.2009.11.006

Begossi, A. 2010. Small-scale fisheries in Latin America. *Management models and challenges*, 9, (pp. 7–31).

Begotti, R., & Landesmann, L. (2008). Predação de ninhos por um grupo híbrido de sagüis (*Callithrix Jacchus*/Penicillata) Introduzidos em área urbana: implicações para a estrutura da comunidade. *Neotropical,* 15(1), (pp. 28–29) . https://doi.org/10.1896/044.015.0107

Bellard, C., Leclerc, C., & Courchamp, F. (2014). Impact of sea level rise on the 10 insular biodiversity hotspots. *Global Ecology and Biogeography*, 23(2), (pp. 203-212).

Bellard, C., Thuiller, W., Leroy, B., Genovesi, P., Bakkenes, M., & Courchamp, F. (2013). Will climate change promote future invasions? *Global Change Biology*, 19(12), (pp. 3740–3748).

Belnap J., & Büdel B. (2016) Biological soil crusts as soil stabilizers. In: Weber B., Büdel B., & Belnap J. (eds) Biological soil crusts: an organizing principle in drylands. *Ecological Studies (Analysis and Synthesis),* vol 226. (pp. 305–320) Springer, Cham. <http://link.springer.com/10.1007/978-3-319-30214-0_16>. Accessed 5 Jun 2016.

Benayas, R. J. M., Newton, A. C., Diaz, A., & Bullock, J. M. (2009). Enhancement of biodiversity and ecosystem services by ecological restoration: a meta-analysis. *Science*, (New York, N.Y.) 325, (pp. 1121–4). <http://www.ncbi.nlm.nih.gov/pubmed/19644076>.

Benson, A., Raikow, D., Larson, J., Fusaro, A., & Bogdanoff, A. (2017). *Dreissena polymorpha* (Pallas, 1771) USGS Nonindigenous Aquatic Species Database, Gainesville, FL.

Berg, Jr C. J., & Olsen, D. A. (1989). Conservation and management of queen conch (*Strombus gigas*) fisheries in the Caribbean. In: Caddy, J.F. (ed.), Marine invertebrate fisheries: Their assessment and management, (pp. 421-442). John Wiley, NY.

Bergan, T., Gallardo, L., & Rodhe, H. (1999). Mercury in the global troposphere: A three-dimensional model study. *Atmospheric Environment*, 33(10), (pp. 1575–1585). https://doi.org/10.1016/S1352-2310(98)00370-7

Berkes, F. (2007). Community-based conservation in a globalized world. *Proceedings of the National academy of sciences*, 104(39), (pp. 15188-15193).

Betancur-R., R., Hines, A., Acero P., A., Ortí, G., Wilbur, A. E., & Freshwater, D. W. (2011). Reconstructing the lionfish invasion: insights into Greater Caribbean biogeography. *Journal of Biogeography*, 38(7), (pp. 1281–1293). https://doi.org/10.1111/j.1365-2699.2011.02496.x

Beyer, W.N., Franson, J.C., French, J.B., May, T., Rattner, B.A., Shearn-Bochsler, V.I., Warner, S.E., Weber, J., & Mosby, D. (2013). Toxic exposure of songbirds to lead in the Southeast Missouri Lead Mining District. *Archives of environmental contamination and toxicology*, 65(3), (pp.598-610).

Bhattacharya, A., Romani, M., & Stern, N. (2012). Infrastructure for development: meeting the challenge. In: Centre for Climate Change Economics and Policy, vol. 15. www.cccep.ac.uk/Publications/Policy/docs/PP-infrastructure-for-development-meeting-the-challenge.pdf.

Bigatti G., & Penchaszadeh P.E. (2008). Invertebrados del Mar Patagónico, diagnóstico de la problemática actual y potencial de su conservación y manejo. In: Estado de Conservación del Mar Patagónico y áreas de influencia [on-line]. Puerto Madryn, Argentina: Edición del Foro. (pp 105–133). http://www.marpatagonico. org/libro/articulo.php?id = bigatti-penchaszdeh-invertebrados.

Bilbao, B. A., Leal, A. V., & Méndez, C. L. (2010). Indigenous use of fire and forest loss in Canaima National Park, Venezuela. Assessment of and tools for alternative strategies of fire management in Pemón indigenous lands. *Human Ecology*, 38(5), (pp. 663-673).

Bingham, H., Fitzsimons, J. A., Redford, K. H., Brent, A., Bezaury-creel, J., & Cumming, T. L. (2017). Privately protected areas: advances and challenges in guidance, policy and documentation. *Parks*, 23(1).

Bjerregaard, P., & Hansen, J.C. (2000). Organochlorines and heavy metals in pregnant women from the Disko Bay area in Greenland. *Science of the total environment*, 245(1), (pp.195-202).

Blackburn, T. M., Essl, F., Evans, T., Hulme, P. E., Jeschke, J. M., Kühn, I., Kumschick, S., Marková, Z., Mrugała, A., Nentwig, W., Pergl, J., Pyšek, P., Rabitsch, W., Ricciardi, A., Richardson, D. M., Sendek, A., Vilà, M., Wilson, J. R. U., Winter, M., Genovesi, P., & Bacher, S. (2014). A Unified Classification of Alien

Species Based on the Magnitude of their Environmental Impacts. *PLoS Biology*, 12(5), e1001850. https://doi.org/10.1371/journal.pbio.1001850

Blackwell, B.D., & Driscoll, C.T. (2015). Deposition of mercury in forests along a montane elevation gradient. *Environmental science & technology*, 49(9), (pp.5363-5370).

Blaser, J., & Zabel, A. (2015). Forest crime in the tropics. *Tropical forestry handbook*. Springer, Berlin, Heidelberg.

Bobbink, R., Hicks, K., Galloway, J., Spranger, T., Alkemade, R., Ashmore, M., Bustamante, M., Cinderby, S., Davidson, E., Dentener, F. and Emmett, B. (2010). Global assessment of nitrogen deposition effects on terrestrial plant diversity: a synthesis. *Ecological applications*, 20(1), (pp.30-59).

Bohlen, P. P. J., Scheu, S., Hale, C. M., McLean, M. A., Migge, S., Groffman, P. M., & Parkinson, D. (2004). Non-native invasive earthworms as agents of change in northern temperate forests. *Frontiers in Ecology and the Environment*, 2(8), (pp. 427–435). https://doi.org/10.1890/1540-9295(2004)002[0427:NIEAAO]2.0.CO;2

Boisier, J. P., Ciais, P., Ducharne, A., & Guimberteau, M. (2015). Projected strengthening of Amazonian dry season by constrained climate model simulations. *Nature Climate Change* 5, (pp.656–660). <http://www.scopus.com/inward/record.url?eid=2-s2.0-84932110744&partnerID=tZOtx3y1>.

Boltovskoty, D., & N. Correa. (2014). Ecosystem impacts of the invasive bivalve *Limnoperna fortunei* (golden mussel) in South America. *Hydrobiologia*,746 (1), (pp. 81-95). DOI 10.1007/s10750-014-1882-9.

Boltovskoy, D. (ed.). (2015). Larval Development of *Limnoperna fortunei*. In: D. Boltovskoy (Ed.), *Limnoperna fortunei*: The ecology, distribution and control of a swiftly spreading invasive fouling mussel (pp. 43–53). Cham: Springer International Publishing.

Boltovskoy, D. (ed.). (2015). *Limnoperna fortunei*, Invading nature - springer Series in Invasion Ecology 10, DOI 10.1007/978-3-319-13494-9_2

Bond, N. A., Cronin, M. F., Freeland, H., & Mantua, N. (2015). Causes and impacts of the 2014 warm anomaly in the NE Pacific. *Geophysical Research Letters*, 42(9), (pp. 3414-3420).

Bond, W. J. (2016). Ancient grasslands at risk. *Science*, 351(6269), (pp.120-122).

Booth, S., & Zeller, D. (2005). Mercury, food webs, and marine mammals: implications of diet and climate change for human health. *Environmental Health Perspectives*, 113(5), (pp. 521-526).

Boretto, J. G., Pacher, N., Giunta, D., Gallucci, G. L., Alfie, V., & De Carli, P. (2014). Comparative clinical study of locking screws versus smooth locking pegs in volar plating of distal radius fractures. *Journal of Hand Surgery* (European Volume), 39(7), (pp.755-760).

Borrvall, C., & Ebenman, B. (2006). Early onset of secondary extinctions in ecological communities following the loss of top predators. *Ecology Letters,* 9, (pp. 435–442).

Bowerman, W. W., Best, D. A., Grubb, T. G., Zimmerman, G. M., & Giesy, J. P. (1998). Trends of contaminants and effects in bald eagles of the Great Lakes Basin. *Environmental Monitoring and Assessment,* 53, (pp.197–212).

Bowerman, W. W., Giesy, J. R., Best, D. A., & Kramer, V. J. (1995). A review of factors affecting productivity of bald eagles in the great lakes region : implications for recovery. *Environmental health perspectives*, 103, (pp. 51–59).

Bramanti, L., Movilla, J., Guron, M., Calvo, E., Gori, A., Dominguez-Carrió, C., Grinyó, J., Lopez-Sanz, A., Martinez-Quintana, A., Pelejero, C., Ziveri, P., & Rossi, S. (2013). Detrimental effects of ocean acidification on the economically important Mediterranean red coral (*Corallium rubrum*). *Global Change Biology,* 19, (pp.1897–1908).

Brancalion, P. H. S., Viani, R. A. G., Rodrigues, R. R., & Gandolfi, S. (2012). Avaliação e monitoramento de áreas em processo de restauração. Martins, *SV Restauração ecológica de ecossistemas degradados*, 1, (pp. 262-293).

Brandeis, T. J., Helmer, E. H., Marcano-Vega, H., & Lugo, A. E. (2009). Climate shapes the novel plant communities that form after deforestation in Puerto Rico and the U.S. Virgin Islands. *Forest Ecology and Management*, 258(7), (pp. 1704–1718). https://doi.org/10.1016/j.foreco.2009.07.030

Brashares, J. S., Arcese, P., Sam, M. K., Coppolillo, P. B., Sinclair, A. R. E., & Balmford, A. (2004). Bushmeat hunting, wildlife declines, and fish supply in West Africa. *Science*, 1180, (pp.1180–1183).

Brauman, K. A., Richter, B. D., Postel, S., Malsy, M., & Flörke, M. (2015). Water depletion : An improved metric for incorporating seasonal and dry-year water scarcity into water risk assessments. 1–12.

Braune, B. M., Outridge, P. M., Fisk, A., Muir, D. C. G., Helm, P. A., Hobbs, K., Hoekstra, P. F., Kuzyk, Z.A., Kwan, M., Letcher, R. J., & Lockhart, W.L. (2005). Persistent organic pollutants and mercury in marine biota of the Canadian Arctic: an overview of spatial and temporal trends. *Science of the Total Environment*, 351, (pp.4-56).

Brennan, L. A., & Kuvlesky Jr, W. P. (2005). North American grassland birds: an unfolding conservation crisis? *Journal of Wildlife Management*, 69(1), (pp. 1-13).

Broad, S., Mulliken, T., & Roe, D. (2003). The nature and extent of legal and illegal trade in wildlife. (pp. 3–22). In: Oldfield, S. (ed.). The trade in wildlife: regulation for conservation. Earthscan, London.

Brokaw, N. V. (1985). Gap-phase regeneration in a tropical forest. *Ecology*, 66(3), (pp. 682-687).

Brook, B.W., Sodhi, N.S. & Bradshaw, C.J. (2008). Synergies among extinction drivers under global change. *Trends in ecology & evolution*, 23(8), (pp.453-460).

Brookes, G. & Barfoot, P. (2007). Global impact of biotech crops: Socio-economic and environmental effects in the first ten years of commercial use. *The journal of agribiotechnology management & economics*, 9(3).

Brooks & Smith (2001) Caribbean catastrophes. *Science*, 294, (pp.1469–1470). DOI: 10.1126/science.1066927.

Brooks, M. L., & Chambers, J. C. (2011). Resistance to invasion and resilience to fire in desert shrublands of North America. *Rangeland Ecology & Management*, 64(5), (pp. 431–438). https://doi.org/10.2111/REM-D-09-00165.1

Brooks, M. L., Minnich, R. A., & Agee, J. K. (2006). Southeastern Deserts Bioregion. (pp. 391–414). In: Sugihara, N.G., Van Wagtendonk, j. W., Shaffer, K. E., Fites-Kaufman, J., Thoe, A.E. Fire in California's Ecosystems. First edition. University of California Press. http://www.jstor.org/stable/10.1525/j.ctt1pnb25.21.

Brooks, T. M., Mittermeier, R. A., da Fonseca, G. A. B., Gerlach, J., Hoffmann, M., Lamoreux, J. F., Mittermeier, C. G., Pilgrim, J. D., & Rodrigues, A. S. L. (2006). Global Biodiversity Conservation Priorities. *Science*, 313(5783), (pp. 58–61). https://doi.org/10.1126/science.1127609

Brose, U., & Hillebrand, H. (2016). Biodiversity and ecosystem functioning in dynamic landscapes. *Philosophical Transactions of the Royal Society of London. Series B,* Biological Sciences, 371: 20150267.

Brown, D. G., Polsky, C. Bolstad, P., Brody, S. D., Hulse, D., Kroh, R. Loveland, T. R., & Thomson, A. (2014): Ch. 13: Land Use and Land Cover Change. Climate Change Impacts in the United States. In: Melillo, J. M. Richmond, T.T.C., & Yohe, G. W. (eds) .The Third National Climate Assessment. U.S. Global Change Research Program, (pp. 318-332). doi:10.7930/J05Q4T1Q.

Brown, K. A., Scatena, F. N., & Gurevitch, J. (2006). Effects of an invasive tree on community structure and diversity in a tropical forest in Puerto Rico. *Forest Ecology and Management*, 226(1–3), (pp. 145–152). https://doi.org/10.1016/j.foreco.2006.01.031

Bruckner, A. W. (2005). The importance of ornamental reef fish trade in the wider Caribbean. *International Journal of Tropical Biology and Conservation* 53, (pp. 127–138).

Brugnoli, E., Clemente, J., Riestra, G., Boccardi, L. & Borthagaray, A. (2006). Especies acuáticas exóticas en Uruguay: situación, problemática y gestión. In: Menafra, R., Rodríguez, L., Scarabino, F. & Conde, D. (eds.). Bases para la conservación y manejo de la costa uruguaya. *Vida Silvestre Uruguay*. (pp. 351-362).

Brugnoli, E., Giberto, D. A., Lanfranconi, A., Schiariti, A., Aguilera, F., Bremec, C. S., Barrero, G., & Muniz, P. (2013). El gastópodo invasor *Rapana venosa* (Valenciennes 1846) y sus posibles efectos en el sistema costero estuarial del Río de la Plata. In: Problemática de los ambientes costeros (pp. 1–19).

Brugnoli, E., Giberto, D., Lanfranconi, A., Schiariti, A., Aguilera, F., Bremec, C.S., Barrero, G., & Muniz, P. (2014). El gasterópodo invasor *Rapana venosa* (Valenciennes 1846) y sus posibles efectos en el ecosistema costero estuarial del Río de la Plata. In: Nuevas miradas a la problemática de los ambientes costeros. Sur de Brasil, Uruguay y Argentina. Compilador y Revisor, Gosso, C. DIRAC, Facultad de Ciencias, Montevideo. (pp. 211-228).

Bucher, E. H. (1992). The causes of extinction of the passenger pigeon. *Current ornithology* (pp. 1-36). Springer US.

Bulte, E. H., Damania, R., & López, R. (2007). On the gains of committing to inefficiency: Corruption, deforestation and low land productivity in Latin America. *Journal of Environmental Economics and Management*, 54, (pp. 277–295). http://doi.org/10.1016/j.jeem.2007.05.002

Bunn, A. G., & Goetz, S. J. (2006). Trends in satellite-observed circumpolar photosynthetic activity from 1982 to 2003 : The influence of seasonality, cover type , and vegetation density. *Earth Interactions*, 10 (12).

Burger, J., Gochfeld, M., Jeitner, C., Burke, S., Stamm, T., Snigaroff, R., Snigaroff, D., Patrick, R. and Weston, J., 2007. Mercury levels and potential risk from subsistence foods from the Aleutians. *Science of the Total Environment*, 384(1), (pp.93-1059.

Burke, L., Reytar, K., Spalding, M., & Perry, A. (2011). Reefs at risk revisited. Washington, D.C. www.wri.org.

Burnett, A. W., Kirby, M. E., Mullins, H. T., & Patterson, W. P. (2003). Increasing Great Lake-effect snowfall during the twentieth century: A regional response to global warming? *Journal of Climate*, 16, (pp. 3535–3542).

Bush, M. B., Silman, M. R., & Urrego, D. H. (2004). 48,000 years of climate and forest change in a biodiversity hot spot. *Science*, 303(5659), (pp.827-829).

Bush, S. R., Toonen, H., Oosterveer, P., & Mol, A. P. J. (2013). The "devil's triangle" of MSC certification: Balancing credibility, accessibility and continuous improvement. *Marine Policy*, 37(1), (pp. 288–293). http://doi.org/10.1016/j.marpol.2012.05.011

Bushaw-Newton, K. I., & Sellner, K. G. (1999). Harmful algal blooms. NOAA´s state of the coast report. Silver Spring. MD: National Oceanic and Atmospheric Administration.

Bustamante, M. M. C., Martinelli, L. A., Pérez, T., Rasse, R., Ometto, J. P. H. B., Siqueira P., F., Machado L., S. R., & Marquina, S. (2015). Nitrogen management challenges in major watersheds of South America. *Environmental Research Letters* 10, 065007. IOP Publishing. <http://stacks.iop.org/1748-9326/10/i=6/a=065007?key=crossref.879056f1355a6f61ce40d257c1205aed>.

Bustamante, M. M. C., Nardoto, G. B., Pinto, A. S., Resende, J. C. F., Takahashi, F. S. C., & Vieira, L. C. G. (2012). Potential impacts of climate change on biogeochemical functioning of Cerrado ecosystems. *Brazilian Journal of Biology*, 72(3), (pp. 655-671).

Butt, N., Beyer, H.L., Bennett, J.R., Biggs, D., Maggini, R., Mills, M., Renwick, A.R., Seabrook, L.M., & Possingham, H.P. (2013). Biodiversity risks from fossil fuel extraction. *Science*, 342(6157), (pp.425-426).

Bykova, O., Laursen, A., Bostan, V., Bautista, J., & McCarthy, L. (2006). Do zebra mussels (*Dreissena polymorpha*) alter lake water chemistry in a way that favours Microcystis growth? *Science of the Total Environment*, 371(1–3), (pp. 362–372). https://doi.org/10.1016/j.scitotenv.2006.08.022

Byrne, S., Miller, P., Waghiyi, V., Buck, C.L., von Hippel, F.A., & Carpenter, D.O. (2015). Persistent organochlorine pesticide exposure related to a formerly used defense site on St. Lawrence Island, Alaska: data from sentinel fish and human sera. *Journal of Toxicology and Environmental Health*, Part A, 78(15), (pp.976-992).

Cabré, M. F., Solman, S., & Núñez, M. (2016). Regional climate change scenarios over southern South America for future climate (2080-2099) using the MM5 Model. Mean, interannual variability and uncertainties. *Atmósfera*, 29(1), (pp. 35-60).

Cabrera, A.L., & Willink, A. (1980) Biogeografía de América Latina, Serie Biología, Monografía 13. (p. 117). OEA, Washington, DC.

Callaghan, T. V, L. O. Björn, F. S. C. Iii, Y. Chernov, T. R. Christensen, B. Huntley, R. Ims, M. Johansson, D. J. Riedlinger, S. Jonasson, N. Matveyeva, W. Oechel, N. Panikov, and G. Shaver. 2005. Arctic Tundra and Polar Desert Ecosystems.

Calvo-Alvarado, J., McLennan, B., Sánchez-Azofeifa, A., & Garvin, T. (2009). Deforestation and forest restoration in Guanacaste, Costa Rica: Putting conservation policies in context. *Forest Ecology and Management*, 258(6), (pp. 931-940).

Calle, Z., & Murgueitio, E. (2015) Ganaderos aliados de la biodiversidad en el Magdalena Medio. *Carta Fedegan* ,149, (pp. 80-85).

Calle, Z., Murgueitio, E., & Chará J. (2012) Integrating forestry, cattle-ranching and landscape restoration. *Unasylva*, 239 (63), (pp. 31-40).

Cameron, R.D., Smith, W.T., White, R.G., & Griffith, D.B. (2005). Central Arctic caribou and petroleum development: Distributional, nutritional, and reproductive implications. *Arctic*, 58, (pp. 1-9).

Campuzano, F. J., Mateus, M. D., Leitão, P. C., Leitão, P. C., Marín, V. H., Delgado, L. E., Tironi, A., Pierini, J. O., Sampaio, A. F. P., Almeida, P., & Neves, R. J. (2013). Integrated coastal zone management in South America: A look at three contrasting systems. *Ocean and Coastal Management*, 72, (pp. 22–35). https://doi.org/10.1016/j.ocecoaman.2011.08.002

Camus, P., Castro, S., & Jaksic, F. (2008). El conejo europeo en chile: historia de una invasión biológica. Historia, II, (pp. 305–339). https://doi.org/10.4067/S0717-71942008000200001

Canning-Clode, J. (Ed.) (2015). Biological invasions in changing ecosystems vectors, ecological impacts, management and predictions. De Gruyter.

Carabias, J., Sarukhán, J., de la Maza, J., & Galindo, C. (2010). Patrimonio natural de México. Cien casos de éxito. Comisión Nacional para el Conocimiento y Uso de la Biodiversidad, México.

Cardozo, M. (2011). Economic displacement and local attitude towards protected area establishment in the Peruvian Amazon. *Geoforum*, 42(5), (pp. 603–614). http://doi.org/10.1016/j.geoforum.2011.04.008

Carlson, D.L., Vault, D.S.D., & Swackhamer, D.L. (2010). On the rate of decline of persistent organic contaminants in lake trout (*Salvelinus namaycush*) from the Great Lakes, 1970–2003. *Environmental science & technology*, 44(6), (pp.2004-2010).

Carothers, C. (2015). Fisheries privatization, social transitions, and well-being in Kodiak, Alaska. *Marine Policy*, 61, (pp. 313–322). http://doi.org/10.1016/j.marpol.2014.11.019

Carpenter, K. E., Abrar, M., Aeby, G., Aronson, R. B., Banks, S., Bruckner, A., Chiriboga, A., Cortés, J., Delbeek, J. C., DeVantier, L., Edgar, G. J., Edwards, A. J., Fenner, D., Guzmán, H. M., Hoeksema, B. W., Hodgson, G., Johan, O., Licuanan, W. Y., Livingstone, S. R., Lovell, E. R., Moore, J. A., Obura, D. O., Ochavillo, D., Polidoro, B. A., Precht, W. F., Quibilan, M. C., Reboton, C., Richards, Z. T., Rogers, A. D., Sanciangco, J., Sheppard, A., Sheppard, C., Smith, J., Stuart, S., Turak, E., Veron, J. E. N., Wallace, C., Weil, E., & Wood, E. (2008). One-third of reef-building corals face elevated extinction risk from climate change and local impacts. *Science*, 32, (pp. 560–563).

Carpenter, S. R., Bennett, E. M., & Peterson, G. D. (2006). Scenarios for ecosystem services: An overview. *Ecology and Society*, volume 11.

Carpenter, S. R., Stanley, E. H., & Vander Zanden, M. J. (2011). State of the world's freshwater ecosystems: physical, chemical, and biological changes. *Annual review of Environment and Resources*, 36, (pp. 75-99).

Carranza, A., de Mello, C., Ligrone, A., González, S., Píriz, P., & Scarabino, F. (2010). Observations on the invading gastropod *Rapana venosa* in Punta del Este, Maldonado Bay, Uruguay. *Biological Invasions*, 12(5), (pp. 995-998).

Carranza, A., Scarabino, F., & Ortega, L. (2007). Distribution of large benthic gastropods in the Uruguayan continental shelf and Río de la Plata estuary. *Journal of Coastal Research*, 24, (pp. 161-168).

Casas, G., Scrosati, R., & Luz Piriz, M. (2004). The invasive kelp *Undaria pinnatifida* (Phaeophyceae, Laminariales) reduces native seaweed diversity in Nuevo Gulf (Patagonia, Argentina). *Biological Invasions*, 6(4), (pp. 411–416).

Castellanos-Gallindo, G. A., Cantera, J. R., Saint-Paul, U., & Ferrol-Schulte, D. (2014). Threats to mangrove social-ecological systems in the most luxuriant coastal forests of the Neotropics. *Biodiversersity and Conservation,* 24, (701-704).

Castello, L., Arantes, C. C., Mcgrath, D. G., Stewart, D. J., & De Sousa, F. S. (2015). Understanding fishing-induced extinctions in the Amazon. Aquatic Conservation: Marine and Freshwater Ecosystems, 25(5). https://doi.org/10.1002/aqc.2491

Castello, L., McGrath, D. G., Hess, L. L., Coe, M. T., Lefebvre, P. A., Petry, P., Macedo, M. N., Renó, V. F., & Arantes, C. C. (2013a). The vulnerability of Amazon freshwater ecosystems. *Conservation Letters,* 6, (pp. 217–229).

Castello, L., McGrath, D. G., Hess, L. L., Coe, M. T., Lefebvre, P. A., Petry, P., Macedo, M. N., *Renó, V. F., & Arantes, C. C. 2013b. The vulnerability of Amazon freshwater ecosystems. Conservation Letters*, 0 (2013) (pp. 1–13).

Castro, S. A., Figueroa, J. A., Muñoz-Schick, M., & Jaksic, F. M. (2005). Minimum residence time, biogeographical origin, and life cycle as determinants of the geographical extent of naturalized plants in continental Chile. *Diversity and Distributions*, 11(3), (pp. 183–191). https://doi.org/10.1111/j.1366-9516.2005.00145.x

Cavole, L. M., Demko, A.M., Diner, R. E., Giddings, A., Koester, I., Pagniello, C. M. L. S., Paulsen, M.-L., Ramirez-Valdez, A., Schwenck, S. M., Yen, N.K., Zill, M. E., & Franks, P.J.S. (2016). Biological impacts of the 2013–2015 warm-water anomaly in the Northeast Pacific: Winners, losers, and the future. *Oceanography*, 29(2), (pp. 273–285) http://dx.doi.org/10.5670/oceanog.2016.32

Cella-Ribeiro, A., Assakawa, L. F., Torrente-Vilara, G., Zuanon, J., Leite, R. G., Doria, C., & Duponchelle, F. (2015). Temporal and spatial distribution of young *Brachyplatystoma* spp. (Siluriformes: Pimelodidae) along the rapids stretch of the Madeira River (Brazil) before the construction of two hydroelectric dams. *Journal of Fish Biolog,y* 86, (pp.1429–1437).

CEPAL (2014). Panorama Social de América Latina. Comisión Económica para América Latina y el Caribe. Santiago, Chile.

CEPAL (2015). Balance Preliminar de las Economías de América Latina y el Caribe. Comisión Económica para América Latina y el Caribe. Santiago, Chile.

CEPAL (2016). CEPAL apoya visibilidad estadística y participación de los pueblos indígenas en la Agenda 2030. Santiago, Chile, 15 julio 2016.

CEPAL (2016). Balance preliminar de las economías de América Latina y el Caribe, Santiago, Chile.

CEPAL (2017). Balance preliminar de las economías de América Latina y el Caribe, Santiago, Chile.

Cesar, H., Burke, L., & Pet-soede, L. (2003). The economics of worldwide coral reef degradation. cesar environmental economics consulting, Arnhem, and WWF-Netherlands 14:23. <http://eprints.eriub.org/48/>.

Chacon, C. Fostering conservation of key priority sites and rural development in Central America: the role of private protected areas. *Parks*, 15 (2005), (pp. 39-47)

Chai, S. L., Tanner, E., & McLaren, K. (2009). High rates of forest clearance and fragmentation pre-and post-National Park establishment: The case of a Jamaican montane rainforest. *Biological Conservation*, 142(11), (pp. 2484-2492).

Chakalall, B., & Cochrane, K. L. (1997). The queen conch fishery in the Caribbean - an approach to responsible fisheries management. (pp. 531–554) In: *Proceedings of the 49th Gulf and Caribbean Fisheries Institute*.

Chapin, F. S. I., Trainor, S. F., Cochran, P., Huntington, H., Markon, C., McCammon, M., McGuire, A. D., & Serreze. M. (2014). Ch. 22: Alaska. climate change impacts in the U.S. In: The Third National Climate Assessment. (pp. 514–536). http://nca2014.globalchange.gov/report/regions/alaska.

Charlotte, D. R., Yolande, B. N., Cordonnier, S., & Claude, B. (2016). The invasive lionfish, *Pterois volitans*, used as a sentinel species to assess the organochlorine pollution by chlordecone in Guadeloupe (Lesser Antilles). *Marine pollution bulletin*, 107(1), (pp.102-106).

Charvériat, C. (2000). Natural disasters in Latin America and the Caribbean: An overview of risk. Inter-Americas Development Bank. Working paper # 434.

Cheung, W. W. L., Lam, V. W. Y., Sarmiento, J. L., Kearney, K., Watson, R., Zeller, D., & Pauly, D. (2010). Large-scale redistribution of maximum fisheries catch potential in the global ocean under climate change. *Global Change Biology,* 16, (pp. 24–35).

Chinea, J. D., & Agosto, R. (2007). Forests surrounding the Joyuda Lagoon, Puerto Rico: 67 years of change. *Caribbean Journal of Science*, 43(1), (pp. 142-147).

Chinea, J. D., & Helmer, E. H. (2003). Diversity and composition of tropical secondary forests recovering from large-scale clearing: Results from the 1990 inventory in Puerto Rico. *Forest Ecology and Management*, 180(1–3), (pp. 227–240). https://doi.org/10.1016/S0378-1127(02)00565-0

Christensen, J. H., & Christensen, O. B. (2007). A summary of the PRUDENCE model projections of changes in European climate by the end of this century. *Climatic change*, 81, (pp. 7-30).

Cibils, A. F., & Borrelli, P. R. (2005). Grasslands of Patagonia. *Grasslands of the world*, (pp. 121-170).

Clark, J.R., Cole, M., Lindeque, P.K., Fileman, E., Blackford, J., Lewis, C., Lenton, T.M., & Galloway, T.S. (2016). Marine microplastic debris: a targeted plan for understanding and quantifying interactions with marine life. *Frontiers in Ecology and the Environment*, 14(6), (pp.317-324).

Clark, M. L., Aide, T. M., Grau, H. R., & Riner, G. (2010). A scalable approach to mapping annual land cover at 250 m using MODIS time series data: A case study in the Dry Chaco ecoregion of South America. *Remote Sensing of Environment*, 114(11), (pp. 2816-2832).

Clarke, S. C., Harley, S. J., Hoyle, S. D., & Rice, J. S. (2013). Population Trends in Pacific Oceanic Sharks and the Utility of Regulations on Shark Finning. *Conservation Biolog,y* 27, (pp.197–209).

Clements, E. A., & Fernandes B. M. (2013). Land grabbing, agribusiness and the peasantry in Brazil and Mozambique. *Agrarian South: Journal of Political Economy* 2(1), (pp. 41-69).

Clements, W. H., Carlisle, D. M., Lazorchak, J. M., & Johnson, P. C. (2000). Heavy metals structure benthic communities in Colorado mountain streams. *Ecological Applications*, 10, (pp. 626–638).

Clements, W. H., Cherry, D. S., & Cairns, J. (1988). Structural alterations in aquatic insect communities exposed to copper in laboratory streams. *Environmental Toxicology and Chemistry*, 7, (pp. 715–722). http://dx.doi.org/10.1897/1552-8618(1988)7[715:SAIAIC]2.0.CO;2.

Clements, W. H., Vieira, N. K. M., & Sonderegger, D. L. (2010b). Use of ecological thresholds to assess recovery in lotic ecosystems. *Journal of the North American Benthological Society*, 29, (pp. 1017–1023).

Clements, W. H., Vieira, N. K. M., & Church, S. E. (2010a). Quantifying restoration success and recovery in a metal-polluted stream: A 17-year assessment of physicochemical and biological responses. *Journal of Applied Ecology*, 47, (pp. 899–910).

Clements, W. H., Cadmus, P., & Brinkman. S. F. (2013). Responses of aquatic insects to Cu and Zn in stream microcosms: Understanding differences between single species tests and field responses. *Environmental Science and Technology*, 47, (pp. 7506–7513).

Cline, T. J., Bennington, V, & Kitchell, J. F. (2013). Climate change expands the spatial extent and duration of preferred thermal habitat for Lake Superior fishes. *PLoS ONE*, 8 (4) doi:10.1371/journal.pone.0062279

.

Coat, S., Bocquené, G., & Godard, E. (2006). Contamination of some aquatic species with the organochlorine pesticide chlordecone in Martinique. *Aquatic Living Resources*, 19(2), (pp.181-187).

Coat, S., Monti, D., Legendre, P., Bouchon, C., Massat, F., & Lepoint, G. (2011). Organochlorine pollution in tropical rivers (Guadeloupe): role of ecological factors in food web bioaccumulation. *Environmental Pollution*, 159(6), (pp.1692-1701).

Coates, D. (1995). Inland capture fisheries and enhancement: Status, constraints, and prospects for food security. Paper presented at the Government of Japan/FAO International Conference on Sustainable Contribution of Fisheries to Food Security, Kyoto, Japan 4–9 December 19.

Cóbar-Carranza, A. J., García, R. A., Pauchard, A., & Peña, E. (2014). Effect of Pinus contorta invasion on forest fuel properties and its potential implications on the fire regime of Araucaria araucana and Nothofagus antarctica forests. *Biological Invasions*, 16(11), (pp. 2273–2291). https://doi.org/10.1007/s10530-014-0663-8

Colding, J., & Folke, C. (2001). Social taboos:“invisible” systems of local resource management and biological conservation. *Ecological applications*, 11(2), (pp. 584-600).

Cole, D. C., Kearney, J., Sanin, L. H., Leblanc, A., & Weber, J. P. (2004). Blood mercury levels among Ontario anglers and sport-fish eaters. *Environmental Research*, 95(3), (pp.305-314).

Collen, B., Whitton, F., Dyer, E. E., Baillie, J. E. M., Cumberlidge, N., Darwall, W. R. T., Pollock, C., Richman, N. I., Soulsby, A. M., & Böhm, M. (2014). Global patterns of freshwater species diversity, threat and endemism. *Global Ecology and Biogeography*, 23(1), (pp. 40–51). https://doi.org/10.1111/geb.12096

Collymore, J. (2011) Disaster management in the Caribbean: Perspectives on institutional capacity reform and development, *Environmental Hazards*, 10:1, (pp. 6-22)

Compton, J. E., Harrison, J. A., Dennis, R. L., Greaver, T. L., Hill, B. H., Jordan, S. J., Walker, H., & Campbell, H. V. (2011). Ecosystem services altered b y human changes in the nitrogen cycle: a new perspective for US decision making. *Ecology Letters,* 14, (pp. 804–815).

CONABIO. (2014). Sistema de información sobre especies invasoras en México. Comisión Nacional para el Conocimiento y Uso de la Biodiversidad. 23 de febrero de 2016. URL:http://www.biodiversidad.gob.mx/invasoras

CONAGUA -Comisión Nacional del Agua. (2012). Atlas del agua en México ISBN: 978-607-8246-29-8

Concostrina-Zubiri, L., Pescador, D. S., Martínez, I., & Escudero, A. (2014). Climate and small scale factors determine functional diversity shifts of biological soil crusts in Iberian drylands. *Biodiversity and conservation*, 23(7), (pp.1757-1770).

Condit, R. 1998. Ecological implications of changes in drought patterns: Shifts in forest composition in Panama. *Climatic change* 39, (pp. 413–427).

Contreras, L. C. (1986). Bioenergetics and distribution of fossorial Spalacopus cyanus (Rodentia): thermal stress, or cost of burrowing. *Physiological Zoology*, 59(1), (pp. 20-28).

Cook, B. I., & Seager, R. (2013). The response of the North American Monsoon to increased greenhouse gas forcing. *Journal of Geophysical Research: Atmospheres* 118, (pp.1-10), doi:10.1002/jgrd.50111.

Cook, E. R., Woodhouse, C. A., Eakin, C. M., Meko, D. M., & Stahle, D. W. (2004). Long-term aridity changes in the western United States. *Science*, 306(5698), (pp. 1015-1018).

Cooke, S. J., & Cowx, I. G. (2004). Review of the state of the world fishery resources: inland fisheries.

Cooper, T. F., De'ath, G., Fabricius, K. E., & Lough, J. M. (2008). Declining coral calcification in massive Porites in two nearshore regions of the northern Great Barrier Reef. *Global Change Biology*, 14, (pp. 529–538).

Cooper, O.R., Parrish, D.D., Stohl, A., Trainer, M., Nédélec, P., Thouret, V., Cammas, J.P., Oltmans, S.J., Johnson, B.J., Tarasick, D., Leblanc, T., McDermid, I.S., Jaffe, D., Gao, R., Stith, J., Ryerson, T., Aikin, K., Campos, T., Weinheimer, A., & Avery, M.A. (2010). Increasing springtime ozone mixing ratios in the free troposphere over western North America. *Nature*, 463, (pp. 344-348), 10.1038/nature08708

Cordy, P., Veiga, M. M., Salih, I., Al-Saadi, S., Console, S., Garcia, O., Mesa, L. A., Velásquez-López, P. C., & Roeser, M. (2011). Mercury contamination from artisanal gold mining in Antioquia, Colombia: The world's highest per capita mercury pollution. *Science of the Total Environment,* 410-411, (pp. 154–160). Elsevier B.V. http://dx.doi.org/10.1016/j.scitotenv.2011.09.006.

Correa, S. B., Araujo, J. K., Penha, J. M. F., Nunes da Cunha, C, Stevenson, P. R., & Anderson, J. T. (2015). Overfishing disrupts an ancient mutualism between frugivorous fishes and plants in Neotropical wetlands. *Biological Conservation,* 191, (pp. 159–167).

Coscieme, L., Pulselli, F. M., Niccolucci, V., Patrizi, N., & Sutton, P. C. (2016). Accounting for "land-grabbing" from a biocapacity viewpoint. *Science of the Total Environment*, 539, (pp. 551–559). http://doi.org/10.1016/j.scitotenv.2015.09.021

Costa, B. G. B, Soares, T. M., Torres, R. F., & Lacerda, L. D. (2013). Mercury distribution in a mangrove tidal creek affected by intensive shrimp farming. *Bulletin of Environmental Contamination and Toxicology*, 90:, (pp. 537-541).

Costello, C., Gaines, S. D., & Lynham, J. (2008). Can catch shares prevent fisheries collapse? *Science,* 321.5896, (pp. 1678-1681).

Côté, I. M., Darling, E. S., & Brown, C. J. (2016). Interactions among ecosystem stressors and their importance in conservation. *Proceedings of the Royal Society of London. Series B*, *Biological Sciences*, Vol. 283, No. 1824, February (pp. 20152592). The Royal Society.

Côté, I.M, Gree, S.J., Morris, J.A. Jr., Akins, J.L., & Steinke, D. (2013). Diet richness of invasive Indo-Pacific lionfish revealed by DNA barcoding. *Marine Ecology Progress Series,* 472, (pp. 249−256).

Coupe, R. H., & Capel, P. D. (2016). Trends in pesticide use on soybean, corn and cotton since the introduction of major genetically modified crops in the United States. *Pest Management Science* 72, (pp. 1013–1022).

Coupe, R. H., Barlow, J. R. B., & Capel, P. D. (2012). Complexity of human and ecosystem interactions in an agricultural landscape. *Environmental Development,* 4, (pp. 88–104).

Courtney, T., Westfield, I., & Ries, J. B. (2013). CO 2 -induced ocean acidification impairs calcification in the tropical urchin *Echinometra viridis*. *Journal of Experimental Marine Biology and Ecology,* 440, (pp.169–175).

Cox, J. R., Martin-R, M. H., Ibarra-F, F. A., Fourie, J. H., Rethman, N. F. G., & Wilcox, D. G. (1988). The influence of climate and soils on the distribution of four African grasses. *Journal of Range Management*, 41(March), (pp. 127–139). https://doi.org/10.2307/3898948

Cox, O.N., & Clements, W.H. (2013). An integrated assessment of polycyclic aromatic hydrocarbons (PAHs) and benthic macroinvertebrate communities in Isle Royale National Park. *Journal of Great Lakes Research*, 39(1), (pp.74-82).

Crabbé, A., & Leroy, P. (2008). The handbook of environmental policy evaluation. London: Earthscan.

Crema, L. C., Biudes, J. F. V. , & Camargo, A. F. M. (2011). Effect of Urucu oil (Brazilian Amazon) on the biomass of the aquatic macrophyte *Eichhornia crassipes* (Mart.) Solms (Pontederiaceae). *Acta Limnologica Brasiliensia*, 23, (pp. 406–411).

Cross, F.A., Evans, D.W., & Barber, R.T. (2015). Decadal declines of mercury in adult bluefish (1972–2011) from the mid-Atlantic coast of the USA. *Environmental science & technology*, 49(15), (pp.9064-9072).

Cubbage, F., Diaz, D., Yapura, P., & Dube, F. (2010). Impacts of forest management certification in Argentina and Chile. *Forest Policy and Economics*, 12(7), (pp. 497–504). http://doi.org/10.1016/j.forpol.2010.06.004

Cunha, D. D. A., & Ferreira, L. V. (2012). Impacts of the Belo Monte hydroelectric dam construction on pioneer vegetation formations along the Xingu River, Pará State, Brazil. *Brazilian Journal of Botany*, 35(2), (pp. 159-167).

Cusack, D.F., Lee, J.K., McCleery, T.L., & LeCroy, C.S. (2015). Exotic grasses and nitrate enrichment alter soil carbon cycling along an urban–rural tropical forest gradient. *Global change biology*, 21(12), (pp.4481-4496).

Cyr, D., Gauthier, S., Bergeron, Y., & Carcaillet, C. (2009). Forest management is driving the eastern North American boreal forest outside its natural range of variability. *Frontiers in Ecology and the Environment*, 7(10), (pp. 519-524).

D'Antonio, C. M., & Vitousek, P. M. (1992). Biological Invasions by Exotic Grasses, the Grass/Fire Cycle, and Global Change. *Annual Review of Ecology and Systematics*, 23(1), (pp. 63–87). https://doi.org/10.1146/annurev.es.23.110192.000431

Dai, A., Qian, T., Trenberth, K. E., & Milliman, J. D. (2009). Changes in continental freshwater discharge from 1948 to 2004. *Journal of Climate*, 22, (pp.2773–2792).

Dai, Z., Johnson, K. D., Birdsey, R. A., Hernandez-Stefanoni, J. L., & Dupuy, J. M. (2015). Assessing the effect of climate change on carbon sequestration in a Mexican dry forest in the Yucatan Peninsula. *Ecological Complexity,* 24, (pp. 46–56). Elsevier B.V. http://linkinghub.elsevier.com/retrieve/pii/S1476945X15001002.

Dai, A., Trenberth, K. E., & Qian, T. (2004). A Global Dataset of Palmer Drought Severity Index for 1870–2002: Relationship with Soil Moisture and Effects of Surface Warming. *Journal of Hydrometeorolog*y, 5, (pp. 1117–1130).

Daly, G.L., Lei, Y.D., Teixeira, C., Muir, D.C., Castillo, L.E., Jantunen, L.M., & Wania, F., 2007. Organochlorine pesticides in the soils and atmosphere of Costa Rica. *Environmental science & technology*, 41(4), pp.1124-1130.

Dallaire, R., Muckle, G., Rouget, F., Kadhel, P., Bataille, H., Guldner, L., Seurin, S., Chajès, V., Monfort, C, Boucher, O., Pierre Thomé, J., Jacobson, S. W., Multigner, L. , & Cordier S. (2012). Cognitive, visual, and motor development of 7-month-old Guadeloupean infants exposed to chlordecone. *Environmental Research*, 118, (pp. 79–85).

Darrigran, G. & Pastorino, G. (1995). The recent introduction of asiatic bivalve, Limnoperna fortunei (Mytilidae) in to South America. *The Veliger,* 38, (pp. 183-187)

Darrigran, G. (2002). Potential impact of filter-feeding invaders on temperate inland freshwater environments. *Biological Invasions,* 4 (pp.145-156).

Davidson, L.N., Krawchuk, M.A., & Dulvy, N.K. (2016). Why have global shark and ray landings declined: improved management or overfishing? *Fish and Fisheries*, 17(2), (pp.438-458).

Davis, M., Douglas, C., Calcote, R., Cole, K. L., Green Winkler, M., & Flakne, R. (2000). Holocene climate in the Western Great Lakes national park and lakeshores: implications for future climate change. *Conservation Biology,* 14, (pp.968–983).

Day. O. (2009). The impacts of climate change on biodiversity in Caribbean islands: what we know, what we need to know, and building capacity for effective adaptation. CANARI Technical Report, 386, (pp. 28) ISBN 1-890792-12-8 http://www.canari.org/macarthurclimatechange.html

De Castro, F., Hagenboom, B., & Baud, M. (2016). Gobernanza ambiental en América Latina. Buenos Aires: CLASCO.

De Gouw, J.A., Parrish, D.D., Frost, G.J., &Trainer, M. (2014). Reduced emissions of CO2, NOx, and SO2 from US power plants owing to switch from coal to natural gas with combined cycle technology. *Earth's Future*, 2(2), (pp.75-82).

de Oliveria, M.D., Campos, M.C.S., Paolucci, E.M., Mansur, M.C., Hamilton, S.K. 2015. Colonization and Spread of Limnoperna fortunei in South America. In D. Boltovskoy (ed.),Limnoperna fortunei, Invading Nature - Springer Series in Invasion Ecology 10, DOI 10.1007/978-3-319-13494-9_19. Springer International Publishing Switzerland. pp 333-355.

De Groot, R. S., Wilson, M. A., & Boumans, R. M. (2002). A typology for the classification, description and valuation of ecosystem functions, goods and services. *Ecological economics*, 41(3), (pp. 393-408).

de la Barrera, E., & Castellanos, A. (2007). High temperature effects on gas exchange for the invasive buffel grass (*Pennisetum ciliare* [L.] Link). *Weed Biology and Management*, 7(2), (pp. 128–131). https://doi.org/10.1111/j.1445-6664.2007.00248.x

De la Casa, A.C., & Ovando, G.G. (2014). Climate change and its impact on agricultural potential in the central region of Argentina between 1941 and 2010. *Agricultural and forest meteorology*, 195, (pp.1-11).

de Moura, Y. M., Hilker, T., Lyapustin, A. I., Galvão, L. S., dos Santos, J. R., Anderson, L. O., de Sousa, C. H. R., & Arai, E. (2015). Seasonality and drought effects of Amazonian forests observed from multi-angle satellite data. *Remote Sensing of Environment,* 171 (pp. 278–290).

Defenders of Wildlife. (2016). Trends in Wildlife Imports from Latin America Deined Entry into the United States. *Fact Sheet*, Washington DC.

Deines, J. M., Williams, D., Hamlin, Q., & McLachlan, J. S. (2016). Changes in forest composition in Ohio between Euro-American settlement and the present. *The American Midland Naturalist*, 176(2), (pp.247-271).

Delgadillo M.J., Aguilar O.T., & Rodríguez V.D. (1999). Los aspectos económicos y sociales de El Niño. (181-210 pp). In: Magaña Rueda, V. O. (edit.) Los impactos de El Niño en México. Dirección de protección civil, Secretaria de Gobernación, México.

Delmas, M. A., & Young O.R. (eds). (2009). Governance for the Environment. New York, Cambridge University Press..

Delphin, S., Escobedo, F. J., Abd-Elrahman, A., & Cropper, W. (2013). Mapping potential carbon and timber losses from hurricanes using a decision tree and ecosystem services driver model. *Journal of environmental management*, 129, (pp. 599-607).

Denevan, W. M. (2004). Semi-intensive pre-European cultivation and the origins of anthropogenic dark earths in Amazonia. In: Amazonian dark earths: explorations in space and time (pp. 135-143). Springer Berlin, Heidelberg.

Denslow, J. S., & DeWalt, S. J. (2008). Exotic plant invasions in tropical forests: patterns and hypotheses. *Tropical Forest Community Ecology*, (August), (pp. 409–426).

Depew, D. C., Burgess, N. M., & Campbell, L. M. (2013). Spatial patterns of methylmercury risks to common loons and piscivorous fish in Canada. *Environmental Science and Technology*, 47, (pp. 13093–13103).

Després, C., Beuter, A., Richer, F., Poitras, K., Veilleux, A., Ayotte, P., Dewailly, E., Saint-Amour, D., & Muckle, G. (2005). Neuromotor functions in Inuit preschool children exposed to Pb, PCBs, and Hg. *Neurotoxicology and Teratology*, 27(2), (pp.245-257).

Di Minin, E., & Toivonen, T. (2015). Global Protected Area Expansion: Creating More than Paper Parks. *BioScience*, 65(7), (pp. 637–638). http://doi.org/10.1093/biosci/biv064

Diaz, R.J., & Rosenberg, R. (2008). Spreading dead zones and consequences for marine ecosystems. *Science*, 321(5891), (pp.926-929).

Didham, R. K., Tylianakis, J. M., Gemmell, N. J., Rand, T. A., & Ewers, R. M. (2007). Interactive effects of habitat modification and species invasion on native species decline. *Trends in Ecology and Evolution*, 22(9), (pp. 489–496).

Dietz, R., Outridge, P. M., & Hobson, K. A. (2009)Anthropogenic contributions to mercury levels in present-day Arctic animals—a review. *Science of the Total Environment*. Dec 1;407(24), (pp.6120-31).

Dietz, R., Riget, F.F., Boertmann, D., Sonne, C., Olsen, M.T., Fjeldså, J., Falk, K., Kirkegaard, M., Egevang, C., Asmund, G., &Wille, F. (2006). Time trends of mercury in feathers of West Greenland birds of prey during 1851− 2003. *Environmental science & technology*, 40(19), (pp.5911-5916).

Dietz, R., Sonne, C., Basu, N., Braune, B., O'Hara, T., Letcher, R.J., Scheuhammer, T., Andersen, M., Andreasen, C., Andriashek, D., & Asmund, G. (2013). What are the toxicological effects of mercury in Arctic biota? *Science of the Total Environment*, 443, (pp.775-790).

Dodds, W.K., Clements, W.H., Gido, K., Hilderbrand, R.H., King, R.S. (2010). Thresholds, breakpoints, and nonlinearity in freshwaters as related to management. *Journal of the North American Benthological Society*, 29(3), (pp.988-997).

Donat, M. G., Alexander, L. V., Yang, H., Durre, I., Vose, R., Dunn, R. J. H., Willett, K. M., Aguilar, E., Brunet, M., Caesar, J. , Hewitson, B., Jack, C. , Klein Tank, A. M. G., Kruger, A. C., Marengo, J., Peterson, T. C., Renom, M., Oria Rojas, C., Rusticucci, M., Salinger, J., Elrayah, A. S., Sekele, S. S., Srivastava, A. K., Trewin, B., Villarroel, C., Vincent, L. A., Zhai, P., Zhang, X., & Kitching, S. (2013). Updated analyses of temperature and precipitation extreme indices since the beginning of the twentieth century: The HadEX2 dataset. *Journal of Geophysical Research Atmospheres*, 118, (pp. 2098–2118).

Dorcas, M. E., Willson, J. D., Reed, R. N., Snow, R. W., Rochford, M. R., Miller, M. A., Meshaka, W. E., Andreadis, P. T., Mazzotti, F. J., Romagosa, C. M., & Hart, K. M. (2012). Severe mammal declines coincide with proliferation of invasive Burmese pythons in Everglades National Park. *Proceedings of the National Academy of Sciences*, 109(7), (pp. 2418–2422). https://doi.org/10.1073/pnas.1115226109

Dornelas, M. (2010). Disturbance and change in biodiversity. *Philosophical Transactions of the Royal Society of London B: Biological Sciences*, 365(1558), (pp. 3719-3727).

Drew, J. A. (2005). Use of traditional ecological knowledge in marine conservation. *Conservation biology*, 19(4), (pp. 1286-1293).

Driscoll, C. T., Han, Y. J., Chen, C. Y., Evers, D. C., Lambert, K. F., Holsen, T. M., Kamman, N. C., & Munson, R. K. (2007). Mercury contamination in forest and freshwater ecosystems in the northeastern United States. *Bioscience*, 57, (pp.17–28).

Drummond, M. A., & Loveland, T. R. (2010). Land-use pressure and a transition to forest-cover loss in the eastern United States. B*ioScience*, 60(4), (pp. 286-298).

Dudgeon, D., Arthington, A. H., Gessner, M. O., Kawabata, Z., Knowler, D. J., Lévêque, C., Naiman, R. J., Prieur-Richard, A. H., Soto, D., Stiassny, M. L. J., & Sullivan, C. A. (2006). Freshwater biodiversity: importance, threats, status and conservation challenges. *Biological Reviews of the Cambridge Philosophical Society,* 81, (pp. 163–82). http://www.ncbi.nlm.nih.gov/pubmed/16336747.

Dudka, S., & Adriano, D.C. (1997). Environmental Impacts of Metal Ore Mining and Processing: A Review. *Journal of Environmental Quality*, 26, (pp. 590-602), 10.2134/jeq1997.00472425002600030003x

Dudley, N., Jeanrenaud, J.P., & Sullivan, F. (2014). Bad harvest: The timber trade and the degradation of global forests. Routledge,.

Duke, N. C. (2016). Oil spill impacts on mangroves: recommendations for operational planning and action based on a global review. *Marine Pollution Bulletin,* 109, (pp. 700–715).

Duke, N. C., Meynecke, J. 0., Dittmann, A. M., Ellison, A. M., Anger, K., Berger, U., Cannicci, S., Diele, K., Ewel, K. C., Field, C. D., Koedam, N., Lee, S.Y., Marchand, C., Nordhaus, I., & Dahdouh-Guebas, F. (2007). A world without mangroves? *Science*, 317, (pp.41-42).

Dunbar, J.B., Britsch, L.D., & Kemp, III, E.B. (1992). Land Loss Rates: Report 3, Louisiana Coastal Plain, Technical Report GL-90-2, U.S. Army Corps of Engineers District, New Orleans, LA, 28 p.

Durnford, D., Dastoor, A., Figueras-Nieto, D., & Ryjkov, A. (2010). Long range transport of mercury to the Arctic and across Canada. *Atmospheric Chemistry and Physics*, 10(13), (pp.6063-6086).

Dutfield, G. (2003). Intellectual Property Rights and the Life Science Industries: A Twentieth Century History, Burlington, VT: Ashgate.

Dykstra, C. R., Meyer, M. W., Warnke, D. K., Karasov, W. H., Anderson, D. E., Bowerman, IV W. W., & Giesy, J. P. (1998). Low reproductive rates of Lake Superior Bald Eagles: low food delivery rates or environmental contaminants? *Journal of Great Lakes Research,* 24, (pp. 32–44).

Dykstra, C. R., Meyer, M. W., Stromborg, K. L., Warnke, D. K., Bowerman, IV W. W., & Best, D. A. (2001). Association of low reproductive rates and high contaminant levels in bald eagles on Green Bay, Lake Michigan. *Journal of Great Lakes Research,* 27, (pp. 239–251).

Dykstra, C. R., Meyer, M. W., Rasmussen, P. W., & Warnke, D. K. (2005). Contaminant concentrations and reproductive rate of Lake Superior bald eagles, 1989-2001. *Journal of Great Lakes Research,* 31, (pp. 227–235).

Eakin, C. M., Nim C. J., Brainard, R. E., Aubrecht, C. E., Gledhill, D. K., Muller-Karger, F., Mumby, P. J., Skirving, W. J., Strong, A. E., Wang, M., Weeks, S., W., Wentz, F., & Ziskin, D. (2010). Monitoring coral reefs from space. *The Oceanography Society Magazine*, Special Volume: The Future of Oceanography From Space. December 2010. (pp. 119-133).

Early, R., Bradley, B. A., Dukes, J. S., Lawler, J. J., Olden, J. D., Blumenthal, D. M., Gonzalez, P., Grosholz, E. D., Ibañez, I., Miller, L. P., Sorte, C. J. B., & Tatem, A. J. (2016). Global threats from invasive alien species in the twenty-first century and national response capacities. *Nature Communications*, 7. https://doi.org/10.1038/ncomms12485

Ebisu, K., & Bell, M.L. (2012). Airborne PM2. 5 chemical components and low birth weight in the northeastern and mid-Atlantic regions of the United States. *Environmental health perspectives*, 120(12), (pp.1746).

Echeverría, C., Smith-Ramírez, C., Aronson, J., & Barrera-Cataño, J. I. (2015). Good news from Latin America and the Caribbean: national and international restoration networks are moving ahead. *Restoration Ecology*, 23(1), (pp. 1-3).

Economic Commission for Latin America and the Caribbean (ECLAC) (2014). Compacts for Equality Towards a Sustainable Future. Thirty-fifth Session of ECLAC, Lima, 5-9 May 2014

Elliott, J.E., Kirk, D.A., Elliott, K.H., Dorzinsky, J., Lee, S., Inzunza, E.R., Cheng, K.M., Scheuhammer, T., & Shaw, P. (2015). Mercury in forage fish from Mexico and Central America: implications for fish-eating birds. *Archives of environmental contamination and toxicology*, 69(4), (pp.375-389).

Ellis, E. C., Goldewijk, K. K., Siebert, S., Lightman, D., & Ramankutty, N. (2010). Anthropogenic transformation of the biomes, 1700 to 2000. *Global Ecology and Biogeography*, 19, (pp. 589–606).

Ellison, C., Bank, M. S., Clinton, B. D., Colburn, E. A., Elliott, K., Ford, C. R., & Foster, D. R. (2005). Loss of Foundation Species. Consequences for the Structure and Dynamics of Forested Ecosystems. *Fronteers in Ecology and the Environment,* 3(9), (pp. 479-486) https://doi.org/10.2307/3868635

Ellison, J. (2015). Vulnerability assessment of mangroves to climate change and sea-level rise impacts. *Wetlands Ecology and Management,* 23, (pp.115-137).

Elser, J.J., Andersen, T., Baron, J.S., Bergström, A.K., Jansson, M., Kyle, M., Nydick, K.R., Steger, L., & Hessen, D.O., (2009). Shifts in lake N: P stoichiometry and nutrient limitation driven by atmospheric nitrogen deposition. *Science*, 326(5954), (pp.835-837).

Emery, S. M., & Rudgers, J. A. (2009, August). PS 73-168: Evaluating dune restorations in the Great Lakes region. In: The 94th ESA Annual Meeting.

Engstrom, D., & Swain, E. (1997). Recent decline in atmospheric mercury deposition in the upper Midwest. *Environmental Science & Technology,* 31, (pp. 960–967).

Erickson, H.E., Helmer, E.H., Brandeis, T.J., & Lugo, A.E. (2014). Controls on fallen leaf chemistry and forest floor element masses in native and novel forests across a tropical island. *Ecosphere*, 5(4), (pp.1-28).

Erisman, J.W., Sutton, M.A., Galloway, J., Klimont, Z., & Winiwarter, W. (2008). How a century of ammonia synthesis changed the world. *Nature Geoscience*, 1(10), (pp.636).

Essl, F., Mang, T., & Moser, D. (2011). Ancient and recent alien species in temperate forests: steady state and time lags. *Biological Invasions*, 14(7), (pp. 1331–1342).

ETC Group (2008) Who owns nature? Corporate power and the final frontier in the commodification of life, Issue No. 100, November 2008, https://www.panna.org/sites/default/files/etc_WhoOwnsNature.pdf

Evers, D.C., Han, Y.J., Driscoll, C.T., Kamman, N.C., Goodale, M.W., Lambert, K.F., Holsen, T.M., Chen, C.Y., Clair, T.A., & Butler, T. (2007). Biological mercury hotspots in the northeastern United States and southeastern Canada. *AIBS Bulletin*, 57(1), (pp.29-43).

Evers, D.C., Savoy, L.J., DeSorbo, C.R., Yates, D.E., Hanson, W., Taylor, K.M., Siegel, L.S., Cooley, J.H., Bank, M.S., Major, A., & Munney, K. (2008). Adverse effects from environmental mercury loads on breeding common loons. *Ecotoxicology*, 17(2), (pp.69-81).

Fabricius, K. E. (2005). Effects of terrestrial runoff on the ecology of corals and coral reefs: Review and synthesis. *Marine Pollution Bulletin,* 50, (pp.125–146).

Fabry, V. J., Seibel, B. A., Feely, R. A., & Orr, J. C. (2008). Impacts of ocean acidification on marine fauna and ecosystem processes. *ICES Journal Of Marine Sciences,* 65, (pp. 414–432).

Fadini, P. S., & Jardim, W. F. (2001). Is the Negro River Basin (Amazon) impacted by naturally occurring mercury? *Science of the Total Environment*, 275, (pp.71–82).

Fajardo, L., González, V., Nassar, J. M., Lacabana, P., Portillo Q, C. A., Carrasquel, F., & Rodríguez, J. P. (2005). Tropical dry forests of Venezuela: Characterization and current conservation status. *Biotropica*, 37(4), (pp. 531–546). https://doi.org/10.1111/j.1744-7429.2005.00071.x

FAO (1996). Enseñanzas de la revolución verde: hacia una nueva revolución verde. *Documentos ténicos de referencia*. Roma: http://www.fao.org/docrep/003/w2612s/w2612s06.htm

FAO Committee on Fisheries. (2014). Inland Fisheries: Issues, Developments and Needs.

FAO. (2011). Review of the state of world marine fishery resources. FAO Fisheries and Aquaculture Technical Paper.

FAO (2011) Why Invest in Sustainable Mountain Development? Rome, IT, 83pp. (ISBN 978-92-5-107012-3).

FAO, GEF & TerrAfrica (2011) Transboundary agro-ecosystem management project for the Kagera river basin. FAO Publications, Roma IT, (pp. 8).

FAO (2013) FAO Success Stories on Climate Smart Agriculture. FAO Publications, Roma IT, 28pp.

FAO. (2016a). FAOSTAT Database. Food and Agriculture Organization of the United Nations. Cited 14 December 2016. http://faostat3.fao.org/home/E

FAO. 2016b. FishStatJ - software for fishery statistical time series. Cited 14 December 2016. http://www.fao.org/fishery/

FAO. 2016c. The State of World Fisheries and Aquaculture. Food and Agriculture Organization of the United Nations.

Farmer, C. J., Goodrich, L. J., Inzunza, E. R. & Smith, J. P. (2008). Conservation status of North America's birds of prey. *State of North America's Birds of Prey*, (pp. 303–420).

Fasanella, C. C., Dias, A. C. F. , Rigonato, J., Fiore, M. F., Soares Jr, F. L., Melo, I. S., Pizzirani-Kleiner, A. A., van Elsas, J. D., & Andreote, F. D.(2012). The selection exerted by oil contamination on mangrove fungal communities. *Water, Air, & Soil Pollution,* 223:¡, (pp. 4233-4243).

Brett, F, & Côté, I. M. (2015). Do by-catch reduction devices in longline fisheries reduce capture of sharks and rays? A global meta-analysis. *Fish and Fisheries*, 16.2 (pp. 300-309).

Fearnside, P. M. (2005). Brazil's Samuel Dam: Lessons for hydroelectric development policy and the environment in Amazonia. *Environmental Management,* 35, (pp.1–19).

Fearnside, P. M. (2013). Carbon credit for hydroelectric dams as a source of greenhouse-gas emissions: The example of Brazil's Teles Pires Dam. *Mitigation and Adaptation Strategies for Global Chan*ge 18, (pp. 691–699).

Fearnside, P. M. (2015). Amazon dams and waterways: Brazil???s Tapaj??s Basin plans. Ambio 44:426–439.

Feely, R.A., Sabine, C.L., Byrne, R.H., Millero, F.J., Dickson, A.G., Wanninkhof, R., Murata, A., Miller, L.A. and Greeley, D., 2012. Decadal changes in the aragonite and calcite saturation state of the Pacific Ocean. Global Biogeochemical Cycles, 26(3).

Fernández, R. J., & Reynolds, J. F. (2000). Potential growth and drought tolerance of eight desert grasses: lack of a trade-off?. Oecologia, 123(1), 90-98.

Ferraro, P. J., Hanauer, M. M., Miteva, D. A., Nelson, J. L., & Pattanayak, S. K. (2015). Estimating the impacts of conservation on ecosystem services and poverty by integrating modeling and evaluation, 112(24). http://doi.org/10.1073/pnas.1406487112

Ferreira, C. E. L., Luiz, O. J., Floeter, S. R., Lucena, M. B., Barbosa, M. C., Rocha, C. R., & Rocha, L. A. (2015). First record of invasive lionfish (Pterois volitans) for the Brazilian coast. PLoS ONE, 10(4), 1–5.

Ferreira, L. V., D. A. Cunha, P. P. Chaves, D. C. L. Matos, and P. Parolin. 2013. Impacts of hydroelectric dams on alluvial riparian plant communities in eastern Brazilian Amazonian. Anais da Academia Brasileira de Ciencias 85:1013–1023.

Ferriter, A., editor. 1997. Brazilian pepper management plan for Florida: a report from The Florida Exotic Pest Plant Council's Brazilian pepper task force. http://www.fleppc.org/Manage_Plans/schinus.pdf.

Field, C.B., V.R. Barros, D.J. Dokken, K.J. Mach, M.D. Mastrandrea, T.E. Bilir, M. Chatterjee, K.L. Ebi, Y.O. Estrada, R.C. Genova, B. Girma, E.S. Kissel, A.N. Levy, S. MacCracken, P.R. Mastrandrea, and L.L. White (eds.) Summary for policymakers. In: Climate Change 2014: Impacts, Adaptation, and Vulnerability. Part A: Global and Sectoral Aspects. Contribution of Working Group II to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA, pp. 1-32.

Field, J. P., Breshears, D. D., & Whicker, J. J. (2009). Toward a more holistic perspective of soil erosion: why aeolian research needs to explicitly consider fluvial processes and interactions. Aeolian Research, 1(1), 9-1

Finer, M., & Jenkins, C. N. (2012). Proliferation of hydroelectric dams in the Andean Amazon and implications for Andes-Amazon connectivity. Plos one, 7(4), e35126.

Finer, M., Jenkins, C.N., Pimm, S.L., Keane, B. and Ross, C., 2008. Oil and gas projects in the western Amazon: threats to wilderness, biodiversity, and indigenous peoples. PloS one, 3(8), p.e2932.

Fischer, Joern, David J. Abson, Van Butsic, M. Jahi Chappell, Johan Ekroos, Jan Hanspach, Tobias Kuemmerle, Henrik G. Smith, and Henrik Wehrden. "Land sparing versus land sharing: moving forward." Conservation Letters 7, no. 3 (2014): 149-157.

Fisher, B., & Christopher, T. (2007). Poverty and biodiversity: measuring the overlap of human poverty and the biodiversity hotspots. Ecological Economics, 62(1), 93-101.

Fitzgerald, W.F., Engstrom, D.R., Mason, R.P. and Nater, E.A., 1998. The case for atmospheric mercury contamination in remote areas. Environmental science & technology, 32(1), pp.1-7.

Flachsbarth, I., Willaarts, B., Xie, H., Pitois, G., Mueller, N. D., Ringler, C., & Garrido, A. (2015). The role of Latin America's land and water resources for global food security: environmental trade-offs of future food production pathways. PloS one, 10(1), e0116733.

Fleeger, J. W., K. R. Carman, and R. M. Nisbet. 2003. Indirect effects of contaminants in aquatic ecosystems. Science of the Total Environment 317:207–233.

Flinn, K. M., & Vellend, M. (2005). Recovery of forest plant communities in post‐agricultural landscapes. Frontiers in Ecology and the Environment, 3(5), 243-250.

Foden, W. B., Butchart, S. H. M., Stuart, S. N., Vié, J.-C., Akçakaya, H. R., Angulo, A., DeVantier, L. M., Gutsche, A., Turak, E., Cao, L., Donner, S. D., Katariya, V., Bernard, R., Holland, R. A., Hughes, A. F., O'Hanlon, S. E., Garnett, S. T., Şekercioğlu, Ç. H., & Mace, G. M. (2013). Identifying the World's Most Climate Change Vulnerable Species: A Systematic Trait-Based Assessment of all Birds, Amphibians and Corals. PLoS ONE, 8(6), e65427. https://doi.org/10.1371/journal.pone.0065427

Foley, J. A., Asner, G. P., Costa, M. H., Coe, M. T., DeFries, R., Gibbs, H. K., Howard, E. A., Olson, S., Patz, J., Ramankutty, N., & Snyder, P. (2007). Amazonia revealed: Forest degradation and loss of ecosystem goods and services in the Amazon Basin. Frontiers in Ecology and the Environment, 5(1), 25–32. https://doi.org/10.1890/1540-9295(2007)5[25:ARFDAL]2.0.CO;2

Foley, J. A., N. Ramankutty, K. A. Brauman, E. S. Cassidy, J. S. Gerber, M. Johnston, N. D. Mueller, C. O'Connell, D. K. Ray, P. C. West, C. Balzer, E. M. Bennett, S. R. Carpenter, J. Hill, C. Monfreda, S. Polasky, J. Rockström, J. Sheehan, S. Siebert, D. Tilman, D. P. M. Zaks, and C. O'Connell. 2011. Solutions for a cultivated planet. Nature 478:337–42.

Ford, James D., et al. (2016). "Including indigenous knowledge and experience in IPCC assessment reports", in Nature Climate Change. Perspective. Published online: 24 March 2016 | DOI: 10.1038/NCLIMATE2954.

Foster, P. (2001). The potential negative impacts of global climate change on tropical montane cloud forests. Earth-Science Reviews, 55(1), 73-106.

Foure, J., A. Benassy-Quere, L. Fontagne (2012). The Great Shift: Macroeconomic projections for the World Economy at the 2050 Horizon, G-MonD, Paris School of Economics. Working Paper No. 23, February 2012, pp. 70-72.

Fowler, C. (1994) Unnatural Selection: Technology, Politics and Plant Evolution (Yverdon: Gordon and Breach).

Frank, K. T., Petrie, B., Choi, J. S., & Leggett, W. C. (2005). Trophic cascades in a formerly cod-dominated ecosystem. Science, 308(5728), 1621-1623.

Franklin, K. A., Lyons, K., Nagler, P. L., Lampkin, D., Glenn, E. P., Molina-Freaner, F., Markow, T., & Huete, A. R. (2006). Buffelgrass (Pennisetum ciliare) land conversion and productivity in the plains of Sonora, Mexico. Biological Conservation, 127(1), 62–71. https://doi.org/10.1016/j.biocon.2005.07.018

Fraser, B., 2016. Oil in the forest. Science, 353(6300), pp.641-643.

Fraser, D., Coon, T., Prince, M., Dion, R., & Bernatchez, L. (2006). Integrating traditional and evolutionary knowledge in biodiversity conservation: a population level case study. Ecology and Society, 11(2).

Freedman, J.A., Butler, S. E. & D. H. Wahl. 2012. Impacts of Invasive Asian Carps on Native Food Webs. Final Project Report – Illinois-Indiana Sea Grant. Kaskaskia Biological Station, Illinois Natural History Survey, University of Illinois at Urbana-Champaign, 18p.

Freitas, M. A. B., I. C. G. Vieira, A. L. K. M. Albernaz, J. L. L. Magalhães, and A. C. Lees. 2015. Floristic impoverishment of Amazonian floodplain forests managed for açaí fruit production. Forest Ecology and Management 351:20–27. <http://www.sciencedirect.com/science/article/pii/S0378112715002777>.

Friis, C., Nielsen, J. Ø., Otero, I., Haberl, H., Niewöhner, J., & Hostert, P. (2016). From teleconnection to telecoupling: taking stock of an emerging framework in land system science. Journal of Land Use Science, 11(2), 131-153.

Frommel, A.Y., Maneja, R., Lowe, D., Malzahn, A.M., Geffen, A.J., Folkvord, A., Piatkowski, U., Reusch, T.B. and Clemmesen, C., 2012. Severe tissue damage in Atlantic cod larvae under increasing ocean acidification. Nature Climate Change, 2(1), p.42.

Fuentes-Franco, R., F. Giorgi, E. Coppola, E. Pavia, and F. Graef. 2015. Inter-annual variability of precipitation over Southern Mexico and Central America and its relationship to Sea Surface Temperature from RegCM4 CORDEX projections. Climate Dynamics 45:425–440. Springer Berlin Heidelberg. <http://dx.doi.org/10.1007/s00382-014-2258-6>.

Fuentes-Ramírez, A., Pauchard, A., Cavieres, L. A., & García, R. A. (2011). Survival and growth of Acacia dealbata vs. native trees across an invasion front in south-central Chile. Forest Ecology and Management, 261(6), 1003–1009. https://doi.org/10.1016/j.foreco.2010.12.018

Fuentes, N., Pauchard, A., Sánchez, P., Esquivel, J., & Marticorena, A. (2013). A new comprehensive database of alien plant species in Chile based on herbarium records. Biological Invasions, 15(4), 847–858. https://doi.org/10.1007/s10530-012-0334-6

Fujimura, M., Matsuyama, A., Harvard, J.P., Bourdineaud, J.P. and Nakamura, K., 2012. Mercury contamination in humans in Upper Maroni, French Guiana between 2004 and 2009. Bulletin of environmental contamination and toxicology, 88(2), pp.135-139.

Galetti, M., Bovendorp, R. S., Fadini, R. F., Gussoni, C. O. a., Rodrigues, M., Alvarez, A. D., Guimarães Jr, P. R., & Alves, K. (2009). Hyper abundant mesopredators and bird extinction in an Atlantic forest island. Zoologia, 26(2), 288–298. https://doi.org/10.1590/S1984-46702009000200011

Gall, S. C., and R. C. Thompson, 2015, The impact of debris on marine life. Mar. Pollut. Bull. 92:170–179. doi: 10.1016/j.marpolbul.2014.12.041

García-Frapolli, E., Ramos-Fernández, G., Galicia, E., & Serrano, A. (2009). Land Use Policy The complex reality of biodiversity conservation through Natural Protected Area policy: Three cases from the Yucatan Peninsula, Mexico, 26, 715–722. http://doi.org/10.1016/j.landusepol.2008.09.008

García, J., Ventura, M. I., Requena, M., Hernández, A. F., Parrón, T., & Alarcón, R. (2017). Association of reproductive disorders and male congenital anomalies with environmental exposure to endocrine active pesticides. Reproductive Toxicology, 71, 95-100.

Gardner, T. A., Côté, I. M., Gill, J. A., Grant, A., & Watkinson, A. R. (2003). Long-term region-wide declines in Caribbean corals. Science, 301(5635), 958-960.

Garmo, Ø.A., Skjelkvåle, B.L., de Wit, H.A., Colombo, L., Curtis, C., Fölster, J., Hoffmann, A., Hruška, J., Høgåsen, T., Jeffries, D.S. and Keller, W.B., 2014. Trends in surface water chemistry in acidified areas in Europe and North America from 1990 to 2008. Water, Air, & Soil Pollution, 225(3), p.1880.

Gauthier, P. T., W. P. Norwood, E. E. Prepas, and G. G. Pyle. 2014. Metal-PAH mixtures in the aquatic environment: A review of co-toxic mechanisms leading to more-than-additive outcomes. Aquatic Toxicology 154:253–269.

Gavilán-García, I., Santos-Santos, E., Tovar-Gálvez, L.R., Gavilán-García, A., Suárez, S. and Olmos, J., 2008. Mercury speciation in contaminated soils from old mining activities in Mexico using a chemical selective extraction. Journal of the Mexican Chemical Society, 52(4), pp.263-271.

Gellis, A.C., 2013. Factors influencing storm-generated suspended-sediment concentrations and loads in four basins of contrasting land use, humid-tropical Puerto Rico. Catena, 104, pp.39-57.

Giberto, D.A., Bremec, C.S., Schejter, L., Schiariti, A., Mianzan, H. and E. Acha. 2006. The invasive Rapa Whelk Rapana venosa (Valenciennes, 1846): status and potential ecological impacts in the Río de la Plata estuary, Argentina-Uruguay. Journal of Shellfish Research, 25(3): 919-924.

Giberto, D.A., Schiariti, A. and C.S. Bremec. 2011. Diet and daily consumption rates of Rapana venosa (Valenciennes, 1846) (Gastropoda: Muricidae) from the Río de la Plata (Argentina-Uruguay). Journal of Shellfish Research, 30(2): 349-358.

Gilliom, R. J., J. E. Barbash, C. G. Crawford, P. a. Hamilton, J. D. Martin, N. Nakagaki, L. H. Nowell, J. C. Scott, P. E. Stackelberg, G. P. Thelin, and D. M. Wolock. 2006. Pesticides in the Nation's Streams and Groundwater, 1992–2001: U.S Geological Survey. Circular 1291. <http://pubs.usgs.gov/circ/2005/1291/pdf/circ1291.pdf>.

Giorgi, F., and N. Diffenbaugh. 2008. Developing regional climate change scenarios for use in assessment of effects on human health and disease. Climate Research. Volume 36.

Giri, C., E. Ochieng, L. L. Tieszen, Z. Zhu, A. Singh, T. Loveland, J. Masek, and D. Duke. 2011. Status and distribution of mangrove forests of the world using earth observation satellite data. Global Ecology and Biogeography 20:154–159.

Gledhill, D. K., R. Wanninkhof, F. K. Millero, and M. Eakin. 2008. Ocean acidification of the Greater Caribbean Region 1996-2006. Journal of Geophysical Research: Oceans 113:1–11.

Glibert, P. M. (2017). Eutrophication, harmful algae and biodiversity—Challenging paradigms in a world of complex nutrient changes. Marine Pollution Bulletin.

Glibert, P.M., Harrison, J., Heil, C. and Seitzinger, S., 2006. Escalating worldwide use of urea–a global change contributing to coastal eutrophication. Biogeochemistry, 77(3), pp.441-463.

Global Biodiversity Information Facility. (2011). GBIF Position Paper on Data Hosting Infraestructure for Primary Biodiversity Data. Version 1.0.

Global Environmental Outlook GEO 4, UNEP, 2007, United Nations Environmental Programme. ISBN: 978-92-807-2838-5. 540pp.

Global Footprint Network (GFN). http://data.footprintnetwork.org/countryMetrics.html?cn=all&yr=. Date accessed: June 10, 2017.

Gloor, M., J. Barichivich, G. Ziv, R. Brienen, J. Schöngart, and P. Peylin. 2015. Recent Amazon climate as background for possible ongoing Special Section :

Godar, J., Suavet, C., Gardner, T. A., Dawkins, E., & Meyfroidt, P. (2016). Balancing detail and scale in assessing transparency to improve the governance of agricultural commodity supply chains. Environmental Research Letters, 11(3), 035015.

Godt, J. W., Arnal, C. H., Baum, R. L., Brien, D., Coe, J. A., De Mouthe, J., Ellis, W., Graymer, R. W., Harp, E. L., Hillhouse, J. W., Houdre, N., Howell, D. G., Jayko, A. S., Lajoie, K. R., Morrissey, M. M., Ramsey, D. W., Savage, W. Z., Schuster, R. L., Wieczorek, G.F., & Wilson, R. C. (1999). Maps showing locations of damaging landslides caused by El Niño rainstorms, winter season 1997-98, San Francisco Bay region, California. Retrieved from pubs.usgs.gov/mf/1999/mf-2325

Golicher, D. J., Cayuela, L., & Newton, A. C. (2012). Effects of climate change on the potential species richness of Mesoamerican forests. Biotropica, 44(3), 284-293.

Gomez-Salazar, C., F. Trujillo, M. Portocarrero-Aya, and H. Whitehead. 2012. Population, density estimates, and conservation of river dolphins (Inia and Sotalia) in the Amazon and Orinoco river basins. Marine Mammal Science 28:124–153.

Goosem, S. P., & Tucker, N. I. (2013). Repairing the rainforest. Wet tropics management authority.

Goreau, T. J. (1992). Bleaching and reef community change in Jamaica: 1951–1991. American Zoologist, 32(6), 683-695.

Gorenflo, L. J., Romaine, S., Mittermeier, R. A., & Walker-Painemilla, K. (2012). Co-occurrence of linguistic and biological diversity in biodiversity hotspots and high biodiversity wilderness areas. Proceedings of the National Academy of Sciences, 109(21), 8032-8037.

Gorokhovich, Y., Voros, A., Reid, M., & Mignone, E. (2003). Prioritizing Abandoned Coal Mine Reclamation Projects Within the Contiguous United States Using Geographic Information System Extrapolation. Environmental Management, 32, 527-534, 10.1007/s00267-003-3043-1

Graesser, J., Aide, T. M., Grau, H. R., & Ramankutty, N. (2015). Cropland/pastureland dynamics and the slowdown of deforestation in Latin America. Environmental Research Letters, 10(3), 034017.

Grau, H. R., and M. Aide. 2008. Grau and Aide 2008. Ecology And Society 13.

Green, S., Akins, J., Maljkovic, A., & Côté, I. (2012). Invasive Lionfish Drive Atlantic Coral Reed Fish Declines. PLoS ONE, 7(3), e32596. https://doi.org/10.1371/journal.pone.0032596

Green, S., Akins, J.L., Maljkovic, A, and I.M. Coˆ té. 2012. Invasive Lionfish Drive Atlantic Coral Reef Fish Declines. PLoS ONE 7:3 e32596.

Griffiths, R. W., Schloesser, D. W., Leach, J. H., & Kovalak, W. P. (1991). Distribution and Dispersal of the Zebra Mussel (Dreissena polymorpha) in the Great Lakes Region. Canadian Journal of Fisheries and Aquatic Sciences, 48(8), 1381–1388. https://doi.org/10.1139/f91-165

Grogan, J., A. G. Blundell, R. M. Landis, A. Youatt, R. E. Gullison, M. Martinez, R. Kometter, M. Lentini, and R. E. Rice. 2010. Over-harvesting driven by consumer demand leads to population decline: big-leaf mahogany in South America. Conservation Letters 3:12–20.

Guallar, E., Sanz-Gallardo, M.I., Veer, P.V.T., Bode, P., Aro, A., Gómez-Aracena, J., Kark, J.D., Riemersma, R.A., Martín-Moreno, J.M. and Kok, F.J., 2002. Mercury, fish oils, and the risk of myocardial infarction. New England Journal of Medicine, 347(22), pp.1747-1754.

Guatemala Ramsar National Report 2015 http://www.ramsar.org/sites/default/files/documents/library/cop12nrfguatemala_20140903final.pdf

Gudynas, E. (2011). Buen Vivir: Today's tomorrow. Development, 54(4), 441–447. http://doi.org/10.1057/dev.2011.86

Guha-Sapir D, Hoyois Ph., Below.R. Annual Disaster Statistical Review 2013:The Numbers and Trends. Brussels: CRED; 2014

Guha, S., & Bhattacharya, S. (2014). Non-parametric Non-stationary Modeling of Spatio-Temporal Data Through State Space Approach. arXiv preprint arXiv:1405.6531.

Guinea Barrientos, H. E., Swain, A., Wallin, M. B., & Nyberg, L. (2015). Disaster management cooperation in central america: the case of rainfall-induced natural disasters. Geografiska Annaler: Series A, Physical Geography, 97(1), 85-96.

Gutiérrez-Galindo, E.A., Casas-Beltrán, D.A., Muñoz-Barbosa, A., Macías-Zamora, J.V., Segovia-Zavala, J.A., Orozco-Borbon, M.V. and Daessle, L.W., 2007. Spatial distribution and enrichment of mercury in surface sediments off the northwest coast of Baja California, Mexico. Ciencias Marinas, 33(4), pp.473-482.

Haddad, N. M., L. a. Brudvig, J. Clobert, K. F. Davies, A. Gonzalez, R. D. Holt, T. E. Lovejoy, J. O. Sexton, M. P. Austin, C. D. Collins, W. M. Cook, E. I. Damschen, R. M. Ewers, B. L. Foster, C. N. Jenkins, a. J. King, W. F. Laurance, D. J. Levey, C. R. Margules, B. a. Melbourne, a. O. Nicholls, J. L. Orrock, D.-X. Song, and J. R. Townshend. 2015. Habitat fragmentation and its lasting impact on Earth's ecosystems. Science Advances 1:1–9. <http://advances.sciencemag.org/content/1/2/e1500052.abstract>.

Hall-Spencer, J.M., Rodolfo-Metalpa, R., Martin, S., Ransome, E., Fine, M., Turner, S.M., Rowley, S.J., Tedesco, D. and Buia, M.C., 2008. Volcanic carbon dioxide vents show ecosystem effects of ocean acidification. Nature, 454(7200), pp.96-99.

Halpern, B. S. 2009. A Global Map of Human Impact on. 948:948–953.

Halpern, B. S., Selkoe, K. A., Micheli, F., & Kappel, C. V. (2007). Evaluating and ranking the vulnerability of global marine ecosystems to anthropogenic threats. Conservation Biology, 21(5), 1301-1315.

Hansen, A. J., Piekielek, N., Davis, C., Haas, J., Theobald, D. M., Gross, J. E., Monahan, W. B., Olliff, T., & Running, S. W. (2014). Exposure of U.S. National Parks to land use and climate change 1900-2100. Ecological Applications : A Publication of the Ecological Society of America, 24(3), 484–502. Retrieved from http://www.ncbi.nlm.nih.gov/pubmed/24834735

Hansen, M. C., Potapov, P. V., Moore, R., Hancher, M., Turubanova, S. A., Tyukavina, A., Thau, D., Stehman, S. V., Goetz, S. J., Loveland, T. R., Kommareddy, A., Egorov, A., Chini, L., Justice, C. O., & Townshend, J. R. G. (2013). High-resolution global maps of 21st-century forest cover change. Science, 342(6160), 850–853. https://doi.org/10.1126/science.1244693

Hanski, I., G. a Zurita, M. I. Bellocq, and J. Rybicki. 2013. Species-fragmented area relationship. PNAS 110:12715–20.

Hanson, C., Yonavjak, L., Clarke, C., Minnemeyer, S., Boisrobert, L., Leach, A., Schleeweis, K. 2010. Southern Forests for the Future. Washington, DC.

Hanson, C.; Buckingham, K. DeWitt, S. and Laestadius, L. (2015) The Restoration Diagnostic: A Method for Developing Forest Landscape Restoration Strategies by Rapidly Assessing the status of Key Success Factors. WRI, IUCN, Washington, DC, USA, 90pp. (ISBN: 978-1-56973-875-7)

Hao, Y., Strosnider, H., Balluz, L. and Qualters, J.R., 2016. Geographic variation in the association between ambient fine particulate matter (PM2. 5) and term low birth weight in the United States. Environmental health perspectives, 124(2), p.250.

Hardell, S., Tilander, H., Welfinger-Smith, G., Burger, J. and Carpenter, D.O., 2010. Levels of polychlorinated biphenyls (PCBs) and three organochlorine pesticides in fish from the Aleutian Islands of Alaska. PloS one, 5(8), p.e12396.

Hardin, Garrett. "The Tragedy of the Commons'(1968) 162." Science 1243 (1968).

Harding, J.M. and Mann, R. 1999. Observations on the biology of the Veined Rapa whelk, Rapana venosa (Valenciennes, 1846) in the Chesapeake Bay. Journal of Shellfish Research, 18(1): 9-18.

Hare, J. A., Alexander, M. A., Fogarty, M. J., Williams, E. H., & Scott, J. D. (2010). Forecasting the dynamics of a coastal fishery species using a coupled climate–population model. Ecological Applications, 20(2), 452–464. https://doi.org/10.1890/08-1863.1

Harmon, D., & Loh, J. (2010). The index of linguistic diversity: A new quantitative measure of trends in the status of the world's languages.

Hayhoe, K., J. VanDorn, T. Croley, N. Schlegal, and D. Wuebbles. 2010. Regional climate change projections for Chicago and the US Great Lakes. Journal of Great Lakes Research 36:7–21.

Hebblewhite, M. (2017). Billion-dollar boreal woodland caribou and the biodiversity impacts of the global oil and gas industry. Biological Conservation, 206, 102–111.

Heithaus, M. R., A. Frid, A. J. Wirsing, and B. Worm. 2008. Predicting ecological consequences of marine top predator declines. Trends in Ecology & Evolution 23:202–210.

Helmer, E. H., T. A. Kennaway, D. H. Pedreros, M. L. Clark, H. Marcano-Vega, L. L. Tieszen, T. R. Ruzycki, S. R. Schill, and C. M. S. Carrington. 2008. Land cover and forest formation distributions for St. Kitts, Nevis, St. Eustatius, Grenada and Barbados from decision tree classification of cloud-cleared satellite imagery. Caribbean Journal of Science 44:175–198.

Helmer, E. H., T. J. Brandeis, A. E. Lugo, and T. Kennaway. 2008. Factors influencing spatial pattern in tropical forest clearance and stand age: Implications for carbon storage and species diversity. Journal of Geophysical Research: Biogeosciences 113.

Helmer, E.H., Ruzycki, T.S., Benner, J., Voggesser, S.M., Scobie, B.P., Park, C., Fanning, D.W. and Ramnarine, S., 2012. Detailed maps of tropical forest types are within reach: Forest tree communities for Trinidad and Tobago mapped with multiseason Landsat and multiseason fine-resolution imagery. Forest Ecology and Management, 279, pp.147-166.

Henders, S., Persson, U. M., & Kastner, T. (2015). Trading forests: land-use change and carbon emissions embodied in production and exports of forest-risk commodities. Environmental Research Letters, 10(12), 125012.

Henson, Joshua I., Frank Muller-Karger, Doug Wilson, Steven L. Morey, George A. Maul, Mark Luther and Christine Kranenburg. 2006. Strategic Geographic Positioning of Sea Level Gauges to Aid in Early Detection of Tsunamis in the Intra-Americas Sea. Science of Tsunami Hazards. Vol. 25, No. 3. pages 173-207.

Hernández, D. L., D. M. Vallano, E. S. Zavaleta, Z. Tzankova, J. R. Pasari, S. Weiss, P. C. Selmants, and C. Morozumi. 2016. Nitrogen Pollution Is Linked to US Listed Species Declines. BioScience 66:213–222.

Hernández, G.; Lahmann, E.; Pérez-Gil, R. (2002) Invasores en Mesoamérica y El Caribe (Invasives in Mesoamerica and the Caribbean). -- 1. ed. -- San José, C.R. : UICN,. http://www.issg.org/pdf/publications/GISP/Resources/Mesoamerica.pdf

Herrmann, T. M. (2006). Indigenous knowledge and management of Araucaria araucana forest in the Chilean Andes: implications for native forest conservation. Biodiversity and Conservation, 15(2), 647-662.

Heyman, W. D., Graham, R. T., Kjerfve, B., & Johannes, R. E. (2001). Whale sharks Rhincodon typus aggregate to feed on fish spawn in Belize. Marine Ecology Progress Series, 215, 275-282.

Hietz, P., Turner, B.L., Wanek, W., Richter, A., Nock, C.A. and Wright, S.J., 2011. Long-term change in the nitrogen cycle of tropical forests. Science, 334(6056), pp.664-666.

Hilborn, R. and Ovando, D., 2014. Reflections on the success of traditional fisheries management. ICES Journal of Marine Science, 71(5), pp.1040-1046.

Hill, R., Dyer, G. A., Lozada-Ellison, L. M., Gimona, A., Martín-Ortega, J., Munoz-Rojas, J., & Gordon, I. J. (2015). A social–ecological systems analysis of impediments to delivery of the Aichi 2020 Targets and potentially more effective pathways to the conservation of biodiversity. Global Environmental Change, 34, 22-34.

Hillstrom, M. L., and R. L. Lindroth. 2008. Elevated atmospheric carbon dioxide and ozone alter forest insect abundance and community composition. Insect Conservation and Diversity 1:233–241. <<Go to ISI>://000262759300006>.

Hinsley, A., D. Verissimo, and D. L. Roberts. 2015. Heterogeneity in consumer preferences for orchids in international trade and the potential for the use of market research methods to study demand for wildlife. Biological Conservation 190:80–86.

Hinzman, L. D., N. D. Bettez, W. R. Bolton, F. S. Chapin, M. B. Dyurgerov, C. L. Fastie, B. Griffith, R. D. Hollister, A. Hope, H. P. Huntington, A. M. Jensen, G. J. Jia, T. Jorgenson, D. L. Kane, D. R. Klein, G. Kofinas, A. H. Lynch, A. H. Lloyd, A. D. Mcguire, F. E. Nelson, W. C. Oechel, T. E. Osterkamp, C. H. Racine, V. E. Romanovsky, R. S. Stone, D. A. Stow, M. Sturm, C. E. Tweedie, G. L. Vourlitis, M. D. Walker, D. A. Walker, P. J. Webber, J. M. Welker, K. S. Winker, and K. Yoshikawa. 2005. EVIDENCE AND IMPLICATIONS OF RECENT CLIMATE CHANGE IN NORTHERN ALASKA AND OTHER ARCTIC REGIONS A recent synthesis of evidence from marine , terrestrial and atmospheric stud- ies shows that the climate of the Arctic has warmed significantly in the last 30 . 251–298.

Hixon, M. A., Green, S. J., Albins, M. A., Akins, J. L., & Morris, J. A. (2016). Lionfish: A major marine invasion. Marine Ecology Progress Series, 558(November), 161–165.

Hobbs, R. J. (2000). Land-use changes and invasions. In Invasive species in a changing world (pp. 55–64).

Hodgson, D. L. (2002). Introduction: Comparative perspectives on the indigenous rights movement in Africa and the Americas. American Anthropologist, 104(4), 1037-1049.

Hoekstra JM, Molnar JL, Jennings M, Revenga C, Spalding MD, Boucher TM, Robertson JC, Heibel TJ, Ellison K (2010) The Atlas of Global Conservation: Changes, Challenges, and Opportunities to Make a Difference (ed. Molnar JL). Berkeley: University of California Press.

Hoekstra, J. M., T. M. Boucher, T. H. Ricketts, and C. Roberts. 2005. Confronting a biome crisis: Global disparities of habitat loss and protection. Ecology Letters 8:23–29.

Holcombe, E., Smith, S., Wright, E. et al. Nat Hazards (2012) 61: 351.

Holway, D. A., Lach, L., Suarez, A. V., Tsutsui, N. D., & Case, T. J. (2002). The Causes and Consequences of Ant Invasions. Annual Review of Ecology and Systematics, 33(1), 181–233.

Honduras Ramsar National Report 2015 http://www.ramsar.org/sites/default/files/documents/library/cop12_nr_honduras.pdf

Hooper, M.. P. Mineau, M. E. Zaccagnini and B. Woodbridge (2002). Pesticides and International Migratory Bird Conservation. Chapter 25, Pag. 737-753, in D.J. Hoffman, B.A. Rattnes, G.A. Burton and J. Cairns Jr. (Eds). Handbook of Ecotoxicology. Lewis Publishers, CRC Press, Boca Raton. FloridaHoover, E., Cook, K., Plain, R., Sanchez, K., Waghiyi, V., Miller, P., Dufault, R., Sislin, C. and Carpenter, D.O., 2012. Indigenous peoples of North America: environmental exposures and reproductive justice. Environmental Health Perspectives, 120(12), p.1645.

Hoover, E., Cook, K., Plain, R., Sanchez, K., Waghiyi, V., Miller, P., Dufault, R., Sislin, C., and Carpenter, D, (2012). Indigenous Peoples of North America: Environmental Exposures and Reproductive Justice. Environ Health Perspect, Vol 12, issue 12, 1645-49. DOI:10.1289/ehp.1205422.

Horowitz, H.M., Jacob, D.J., Amos, H.M., Streets, D.G. and Sunderland, E.M., 2014. Historical mercury releases from commercial products: Global environmental implications. Environmental science & technology, 48(17), pp.10242-10250.

Howarth, R.W., 2014. A bridge to nowhere: methane emissions and the greenhouse gas footprint of natural gas. Energy Science & Engineering, 2(2), pp.47-60.

Howell Rivero, L., & Rivas, L. R. (1940). Algunas consideraciones sobre los ciclidos de Cuba. Mem Soc. Cub. Hist. Nat, 14(4), 373-395

Huffman, M. (2013). The many elements of traditional fire knowledge: synthesis, classification, and aids to cross-cultural problem solving in fire-dependent systems around the world. Ecology and Society, 18(4).

Hufkens, K., Keenan, T. F., Flanagan, L. B., Scott, R. L., Bernacchi, C. J., Joo, E., Brunsell, N. A., Verfaillie, J., & Richardson, A. D. (2016). Productivity of North American grasslands is increased under future climate scenarios despite rising aridity. Nature Climate Change, 6(7), 710–714. https://doi.org/10.1038/nclimate2942

Hughes, R. M., Amezcua, F., Chambers, D. M., Daniel, W. M., Franks, J. S., Franzin, W., MacDonald, D., Merriam, E., Neall, G., dos Santos Pompeu, P., Reynolds, L., & Woody, C. A. (2016). AFS Position Paper and Policy on Mining and Fossil Fuel Extraction. Fisheries, 41(1), 12–15. https://doi.org/10.1080/03632415.2016.1121742

Hughes, T. P. (1994). Catastrophes, phase shifts, and large-scale degradation of a Caribbean coral reef. Science-AAAS-Weekly Paper Edition, 265(5178), 1547-1551.

Hung, H., Blanchard, P., Halsall, C.J., Bidleman, T.F., Stern, G.A., Fellin, P., Muir, D.C.G., Barrie, L.A., Jantunen, L.M., Helm, P.A. and Ma, J., 2005. Temporal and spatial variabilities of atmospheric polychlorinated biphenyls (PCBs), organochlorine (OC) pesticides and polycyclic aromatic hydrocarbons (PAHs) in the Canadian Arctic: Results from a decade of monitoring. Science of the Total Environment, 342(1), pp.119-144.

IAWG, U. (2013). Technical support document: Technical update of the social cost of carbon for regulatory impact analysis under executive order 12866. Interagency Working Group on Social Cost of Carbon, United States Government, Washington, DC.

IEA. (2015). Key World Energy Statistics. International Energy Agency, Paris: OECD/IEA.

IEA. (2016). Key World Energy Statistics. International Energy Agency, Paris: OECD/IEA.

Ibañez, M., & Blackman, A. (2016). Is Eco-Certification a Win-Win for Developing Country Agriculture? Organic Coffee Certification in Colombia. World Development, 82, 14-27.

International Monetary Fund (IMF). 2014. World Economic Outlook: Legacies, Clouds, Uncertainties. Washington (October). www.imf.org

International Monetary Fund (IMF). (2015). World Economic Outlook: Adjusting to Lower Commodity Prices. Washington (October). www.imf.org

International Monetary Fund (IMF). (2016). World Economic Outlook: Subdued Demand: Symptoms and Remedies. Washington, October. www.imf.org

International Monetary Fund (IMF). 2017. Seeking Sustainable Growth: Short-Term Recovery, Long-Term Challenges. Washington, DC, October. www.imf.org

International Tropical Timber Organization (ITTO). (2011). 25 Success stories - Illustrating ITTO's 25-year quest to sustain tropical forests. Yokohama, Japan, 62pp. (ISBN 4-902045-89-3)

Inoue, C. Y. A., & Moreira, P. F. (2016). Many worlds, many nature(s), one planet: indigenous knowledge in the Anthropocene. Revista Brasileira de Política Internacional, 59(2), 1–19.

Intergovernmental Panel on Climate Change. 2014. Climate Change 2014 Synthesis Report Summary Chapter for Policymakers. IPCC.

IPBES (2016). The methodological assessment report on scenarios and models of biodiversity and ecosystem services. Ferrier, S., Ninan, K. N., Leadley, P., Alkemade, R., Acosta, L. A., Akçakaya, H. R., Brotons, L., Cheung, W. W. L., Christensen, V., Harhash, K. A., Kabubo-Mariara, J., Lundquist, C., Obersteiner, M., Pereira, H. M., Peterson, G., Pichs-Madruga, R., Ravindranath, N., Rondinini C., & Wintle, B.A. (eds.). Secretariat of the Intergovernmental. Science-Policy Platform on Biodiversity and Ecosystem Services, Bonn, Germany

IPCC, (2014a). Climate Change 2014: Impacts, Adaptation, and Vulnerability. Part A: Global and Sectoral Aspects. Contribution of Working Group II to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change [Field, C.B., V.R. Barros, D.J. Dokken, K.J. Mach, M.D. Mastrandrea, T.E. Bilir, M. Chatterjee, K.L. Ebi, Y.O. Estrada, R.C. Genova, B. Girma, E.S. Kissel, A.N. Levy, S. MacCracken, P.R. Mastrandrea, and L.L.White (eds.)]. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA, 1132 pp.

IPCC, (2014b). Summary for policymakers. In: Climate Change 2014: Impacts, Adaptation, and Vulnerability. Part A: Global and Sectoral Aspects. Contribution of Working Group II to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change [Field, C.B., V.R. Barros, D.J. Dokken, K.J. Mach, M.D. Mastrandrea, T.E. Bilir, M. Chatterjee, K.L. Ebi, Y.O. Estrada, R.C. Genova, B. Girma, E.S. Kissel, A.N. Levy, S. MacCracken, P.R. Mastrandrea, and L.L. White (eds.)]. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA, pp. 1-32.

Iriarte, J. A., Lobos, G. A., & Jaksic, F. M. (2005). Invasive vertebrate species in Chile and their control and monitoring by governmental agencies. Revista Chilena de Historia Natural, 78(1), 143–154.

Irigoyen, A. J., Trobbiani, G., Sgarlatta, M. P., & Raffo, M. P. (2011). Effects of the alien algae Undaria pinnatifida (Phaeophyceae, Laminariales) on the diversity and abundance of benthic macrofauna in Golfo Nuevo (Patagonia, Argentina): Potential implications for local food webs. Biological Invasions, 13(7), 1521–1532.

Isbell, F., D. Tilman, S. Polasky, and M. Loreau. 2015. The biodiversity-dependent ecosystem service debt. Ecology Letters. Volume 18.

Isbell, F., Gonzalez, A., Loreau, M., Cowles, J., Díaz, S., Hector, A., Mace, G. M., Wardle, D. A., O'Connor, M. I., Duffy, J. E., Turnbull, L. A., Thompson, P. L., & Larigauderie, A. (2017). Linking the influence and dependence of people on biodiversity across scales. Nature, 546(7656), 65–72. https://doi.org/10.1038/nature22899

Isbell, F., Reich, P. B., Tilman, D., Hobbie, S. E., Polasky, S., & Binder, S. (2013). Nutrient enrichment, biodiversity loss, and consequent declines in ecosystem productivity. Proceedings of the National Academy of Sciences, 110(29), 11911-11916.

IUCN, (2016). The IUCN Red List of Threatened Species. Version 2016. <www.iucnredlist.org>. Downloaded on 12 May 2016.

Jackson, J. B. C., Donovan, M. K., Cramer, K. L., & Lam, V. V. (2014). Status and trends of Caribbean coral reefs. Gland: Global Coral Reef Monitoring Network, IUCN. Available at http://cmsdata. iucn. org/downloads/caribbean_coral_reefs___status_report_1970_2012. pdf.

Jackson, J. B. C., Kirby, M. X., Berger, W. H., Bjorndal, K. A., Botsford, L. W., Bourque, B. J., Bradbury, R. H., Cooke, R., Erlandson, J., Estes, J. A., Hughes, T. P., Kidwell, S., Lange, C. B., Lenihan, H. S., Pandolfi, J. M., Peterson, C. H., Steneck, R. S., Tegner, M. J., & Warner, R. R. (2001). Historical Overfishing and the Recent Collapse of Coastal Ecosystems. Science, 293(5530), 629 LP-637. Retrieved from http://science.sciencemag.org/content/293/5530/629.abstract

Jacobs, D. F., Dalgleish, H. J., & Nelson, C. D. (2013). A conceptual framework for restoration of threatened plants: The effective model of American chestnut (Castanea dentata) reintroduction. New Phytologist, 197(2), 378–393.

Jambeck, J. R., Geyer, R., Wilcox, C., Siegler, T. R., Perryman, M., Andrady, A., Narayan, R., & Law, K. L. (2015). Marine pollution. Plastic waste inputs from land into the ocean. Science (New York, N.Y.), 347(6223), 768–771. https://doi.org/10.1126/science.1260352

Jantz, S. M., Barker, B., Brooks, T. M., Chini, L. P., Huang, Q., Moore, R. M., Noel, J., & Hurtt, G. C. (2015). Future habitat loss and extinctions driven by land-use change in biodiversity hotspots under four scenarios of climate-change mitigation. Conservation Biology, 29(4), 1122–1131. https://doi.org/10.1111/cobi.12549

Jaramillo, A., Osman, D., Caputo, L. & L. Cardenas . 2015. Molecular evidence of a Didymosphenia geminata (Bacillariophyceae) invasion in Chilean freshwater systems. Harmful Algae 49 117–123

Jaramillo, E., Dugan, J. E., Hubbard, D. M., Melnick, D., Manzano, M., Duarte, C., Campos, C., & Sanchez, R. (2012). Ecological implications of extreme events: Footprints of the 2010 earthquake along the Chilean coast. PLoS ONE, 7(5). https://doi.org/10.1371/journal.pone.0035348

Jaspers, V.L.B., Sonne, C., Soler-Rodriguez, F., Boertmann, D., Dietz, R., Eens, M., Rasmussen, L.M. and Covaci, A., 2013. Persistent organic pollutants and methoxylated polybrominated diphenyl ethers in different tissues of white-tailed eagles (Haliaeetus albicilla) from West Greenland. Environmental pollution, 175, pp.137-146.

Jenkins, C. N., Pimm, S. L., & Joppa, L. N. (2013). Global patterns of terrestrial vertebrate diversity and conservation. Proceedings of the National Academy of Sciences, 110(28), E2602–E2610.

Jeschke, J. M., & Strayer, D. L. (2005). Invasion success of vertebrates in Europe and North America. Proceedings of the National Academy of Sciences of the United States of America, 102(20), 7198-7202.

Jetoo, S., Thorn, A., Friedman, K., Gosman, S., & Krantzberg, G. (2015). Governance and geopolitics as drivers of change in the Great Lakes—St. Lawrence basin. Journal of Great Lakes Research, 41(1), 108-118.

Jia, Y., Yu, G., Gao, Y., He, N., Wang, Q., Jiao, C., & Zuo, Y. (2016). Global inorganic nitrogen dry deposition inferred from ground-and space-based measurements. Scientific reports, 6.

Jiménez, A., Pauchard, A., Cavieres, L. A., Marticorena, A., & Bustamante, R. O. (2008). Do climatically similar regions contain similar alien floras? A comparison between the mediterranean areas of central Chile and California. Journal of Biogeography, 35(4), 614–624.

Johnson, J. W., Oelkers, E. H., & Helgeson, H. C. (1992). SUPCRT92: A software package for calculating the standard molal thermodynamic properties of minerals, gases, aqueous species, and reactions from 1 to 5000 bar and 0 to 1000 C. Computers & Geosciences, 18(7), 899-947.

Johnson, L. E., & Padilla, D. K. (1996). Geogaphic spread of exotic species. Ecological lessons and opportunities from the invasion of the zebra mussel Dreissena polymorpha. Biological Conservation, 78(96), 23–33.

Johnson, W. C., & Poiani, K. A. (2016). Climate Change Effects on Prairie Pothole Wetlands: Findings from a Twenty-five Year Numerical Modeling Project. Wetlands, 36(S2), 273–285. https://doi.org/10.1007/s13157-016-0790-3

Johnstone, J. F., Hollingsworth, T. N., Chapin, F. S., & Mack, M. C. (2010). Changes in fire regime break the legacy lock on successional trajectories in Alaskan boreal forest. Global Change Biology, 16(4), 1281-1295.

Jones, H. P., and O. J. Schmitz. 2009. Rapid recovery of damaged ecosystems. PLoS ONE 4.

Jones, N., McGinlay, J., & Dimitrakopoulos, P. G. (2017). Improving social impact assessment of protected areas: A review of the literature and directions for future research. Environmental Impact Assessment Review, 64, 1–7. http://doi.org/10.1016/j.eiar.2016.12.007

Jørgensen, O. A., Bastardie, F., and Eigaard, O. R. 2014. Impact of deep-sea fishery for Greenland halibut (Reinhardtius hippoglossoides) on noncommercial fish species off West Greenland. – ICES Journal of Marine Science, 71: 845–852.

Joyce, L. A., D. Running, V. Breshears, R. Dale, R. W. Malmsheimer, B. Sampson, B. Sohngen, and C. W. Woodall. 2014. Forests: Climate change impacts in the United States. U.S. Global Change Research Program 175–194. <http://nca2014.globalchange.gov/report/our-changing-climate/introduction>.

Juliano, S. A., & Philip Lounibos, L. (2005). Ecology of invasive mosquitoes: Effects on resident species and on human health. Ecology Letters, 8(5), 558–574.

Junk, W. J. (2013). Current state of knowledge regarding South America wetlands and their future under global climate change. Aquatic Sciences, 75(1), 113-131.

Junk, W. J., M. G. M. Soares, and P. B. Bayley. 2007. Freshwater fishes of the Amazon River basin: their biodiversity, fisheries, and habitats. Aquatic Ecosystem Health & Management 10:153–173. Karatayev, A.Y., Padilla, D., Minchin, D. , Boltovskoy, D. & L.B. Burlakova. 2007. Changes in global economies and trade : the potential spread of exotic freshwater bivalves. Biological Invasions 9: 161-180.

Junk, W. J., M. T. F. Piedade, J. Schöngart, and F. Wittmann. 2012. A classification of major natural habitats of Amazonian white-water river floodplains (várzeas). Wetlands Ecology and Management 20:461–475.

Junk, W. J., M. T. F. Piedade, R. Lourival, F. Wittmann, P. Kandus, L. D. Lacerda, R. L. Bozelli, F. A. Esteves, C. Nunes da Cunha, L. Maltchik, J. Sch??ngart, Y. Schaeffer-Novelli, and A. A. Agostinho. 2014. Brazilian wetlands: Their definition, delineation, and classification for research, sustainable management, and protection. Aquatic Conservation: Marine and Freshwater Ecosystems 24:5–22.

Junqueira, A. B., Shepard, G. H., & Clement, C. R. (2010). Secondary forests on anthropogenic soils in Brazilian Amazonia conserve agrobiodiversity. Biodiversity and Conservation, 19(7), 1933-1961.

Kairo, M., Ali, B., Cheesman, O., Haysom, K., & Murphy, S. (2003). Invasive species threats in the Caribbean region: report to the Nature Conservancy.

Kaiser Family Foundation. (2013). Global health facts – Urban population (Percent of total population living in urban areas). http://kff.org/global-indicator/urban-population/. Accessed 10 July 2013.

Kaplanis, N. J., Harris, J. L., & Smith, J. E. (2016). Distribution patterns of the non-native seaweeds Sargassum horneri (Turner) C. Agardh and Undaria pinnatifida (Harvey) Suringar on the San Diego and Pacific coast of North America. Aquatic Invasions, 11(2), 111–124. https://doi.org/10.3391/ai.2016.11.2.01

Karatayev, A.Y., Padilla, D., Minchin, D. , Boltovskoy, D. & L.B. Burlakova. 2007. Changes in global economies and trade : the potential spread of exotic freshwater bivalves. Biological Invasions 9: 161-180.

Karmalkar, A. V., Bradley, R. S., & Diaz, H. F. (2008). Climate change scenario for Costa Rican montane forests. Geophysical Research Letters, 35(11).

Karmalkar, A. V., R. S. Bradley, and H. F. Diaz. 2011. Climate change in Central America and Mexico: Regional climate model validation and climate change projections. Climate Dynamics 37:605–629.

Kastner, T., Erb, K. H., & Haberl, H. (2015). Global human appropriation of net primary production for biomass consumption in the European Union, 1986–2007. Journal of Industrial Ecology, 19(5), 825-836.

Kauffman, J. B., Trejo, H. H., Garcia, M. D. C. J., Heider, C., & Contreras, W. M. (2016). Carbon stocks of mangroves and losses arising from their conversion to cattle pastures in the Pantanos de Centla, Mexico. Wetlands Ecology and Management, 24(2), 203-216.

Kaufmann, D., Kraay., & A Mastruzzi, M. (2007a). Worldwide Governance Indicators Project: Answering the Critics World Bank Policy Research Working Paper No. 4149. Available at SSRN: https://ssrn.com/abstract=965077

Kawaguchi, S., Ishida, A., King, R., Raymond, B., Waller, N., Constable, A., Nicol, S., Wakita, M. and Ishimatsu, A., 2013. Risk maps for Antarctic krill under projected Southern Ocean acidification. Nature Climate Change, 3(9), p.843.

Keegana, K.M., Mary R. Alberta, Joseph R. McConnellb, and Ian Bakera. Climate change and forest fires synergistically drive widespread melt events of the Greenland Ice Sheet. | PNAS | June 3, 2014 | vol. 111 | no. 22 pp.7964–7967

Keeler B.L., Gourevitch J.D., Polasky S., Isbell F., Tessum C.W., Hill J.D. et al. (2016). The social costs of nitrogen. Science Advances, 2, e1600219

Keenan, R. J., G. A. Reams, F. Achard, J. V. de Freitas, A. Grainger, and E. Lindquist. 2015. Dynamics of global forest area: Results from the FAO Global Forest Resources Assessment 2015. Forest Ecology and Management. Volume 352.

Kerr, L. A., Secor, D. H., & Piccoli, P. M. (2009). Partial migration of fishes as exemplified by the estuarine-dependent white perch. Fisheries, 34(3), 114-123.

Kerckhof F, Vink RJ, Nieweg DC and Post JN. 2006. The veined whelk Rapana venosa has reached the North Sea. Aquatic Invasions, 1: 35-37.

Kershaw, P., Katsuhiko, S., Lee, S. and Woodring, D., 2011. Plastic debris in the ocean. United Nations Environment Programme.

King, R. S., and M. E. Baker. 2010. Considerations for analyzing ecological community thresholds in response to anthropogenic environmental gradients. Journal of the North American Benthological Society 29:998–1008.

Kirby, K. R., Laurance, W. F., Albernaz, A. K., Schroth, G., Fearnside, P. M., Bergen, S., Venticinque, E. M., & da Costa, C. (2006). The future of deforestation in the Brazilian Amazon. Futures, 38(4), 432–453. https://doi.org/10.1016/j.futures.2005.07.011

Kirchner, M., Faus-Kessler, T., Jakobi, G., Levy, W., Henkelmann, B., Bernhöft, S., Kotalik, J., Zsolnay, A., Bassan, R., Belis, C. and Kräuchi, N., 2009. Vertical distribution of organochlorine pesticides in humus along Alpine altitudinal profiles in relation to ambiental parameters. Environmental pollution, 157(12), pp.3238-3247.

Kirwan, M. L., & Megonigal, J. P. (2013). Tidal wetland stability in the face of human impacts and sea-level rise. Nature, 504(7478), 53-60.

Klein E, Cardenas JJ, Esclasans D (2009) Prioridades de conservacio´n de la biodiversidad marina del Frente Atla´ntico y Golfo de Paria.

Knapp, A. K., Blair, J. M., Briggs, J. M., Collins, S. L., Hartnett, D. C., Johnson, L. C., & Towne, E. G. (1999). The keystone role of bison in North American tallgrass prairie: Bison increase habitat heterogeneity and alter a broad array of plant, community, and ecosystem processes. BioScience, 49(1), 39-50.

Knick, S. T., Dobkin, D. S., Rotenberry, J. T., Schroeder, M. A., & Vander Haegen, W. M. (2003). Teetering on the edge or too late? Conservation and research issues for avifauna of sagebrush habitats. The Condor, 105(4), 611-634.

Kocman, D., Horvat, M., Pirrone, N. and Cinnirella, S., 2013. Contribution of contaminated sites to the global mercury budget. Environmental research, 125, pp.160-170.

Koizumi, T. (2015). Biofuels and food security. Renewable and Sustainable Energy Reviews, 52, 829-841.

Kolar, C. S., Chapman, D.C. , Courtenay, W. R. Jr, Housel, Ch. M., Williams, J.D. and D. P. Jennings. 2007. Bigheaded carps: a biological synopsis and environmental risk assessment. American Fisheries Society Special Publication 33. ISBN-13: 978-1-888569-79-7. 204 pp.

Konikow, L. F. 2013. Groundwater Depletion in the United States (1900 – 2008). Scientific Investigations Report 2013 – 5079 75.

Kornis, M. S., Sharma, S., & Jake Vander Zanden, M. (2013). Invasion success and impact of an invasive fish, round goby, in Great Lakes tributaries. Diversity and Distributions, 19(2), 184–198. https://doi.org/10.1111/ddi.12001

Krause-Jensen, D., & Duarte, C. M. (2016). Substantial role of macroalgae in marine carbon sequestration. Nature Geoscience, 9(10), 737-742.

Kreuter, U. P., Iwaasa, A. D., Theodori, G. L., Ansley, R. J., Jackson, R. B., Fraser, L. H., Naeth, M. A., McGillivray, S., & Moya, E. G. (2016, September 15). State of knowledge about energy development impacts on North American rangelands: An integrative approach. Journal of Environmental Management. Academic Press. https://doi.org/10.1016/j.jenvman.2016.05.007

Kunkel, K. E., K. Andsager, and D. D. R. Easterling. 1999. Long-Term Trends in Extreme Precipitation Events over the Conterminous United States and Canada. Journal of Climate 12:2515–2527.

Kunkel, K. E., M. Palecki, L. Ensor, K. G. Hubbard, D. Robinson, K. Redmond, and D. Easterling. 2009. Trends in twentieth-century U.S. snowfall using a quality-controlled dataset. Journal of Atmospheric and Oceanic Technology 26:33–44.

Kurtz, C., & Hansen, M. (2014). An Assessment of Garlic Mustard in Northern U. S. forests., (Nrcs), 1–5.

Lacerda, L. D., and B. Kjerfve. 1999. Conservation and management of latin american mangroves. Pages 183-194 in. W. Salomons, R. K. Turner, L. D. Lacerda, and S. Ramachandran, editors. Perspectives on integrated coastal zone management.

Lacerda, L. D., J. E. Conde, B. Kjerfve, R. Alvarez-León, C. Alarcón, and J. Polanía. 2002. American mangroves. Pages 1-62 in. L. D. Lacerda, editor. Mangrove ecosystems: function and management.

Lacerda, L. D., T. M. Soares, B. G. B. Costa, and M. D. P. Godoy. 2011. Mercury emission factors from intensive shrimp aquaculture and their relative importance to the Jaguaribe River Estuary, NE Brazil. Bulletin of Environmental Contamination and Toxicology 87:657-661.

Lacy, R. C. (2000). Considering threats to the viability of small populations using individual-b. Ecological Bulletins, (48), 39–51.

Lammam, Ch. and H. MacIntyre (2016). An Introduction to the State of Poverty in Canada, Fraser Institute, January 2016: fraserinstitute.org

Lammers, P., Richter, T., Lux, M., Ratsimbazafy, J., & Mantilla-Contreras, J. (2017). The challenges of community-based conservation in developing countries—A case study from Lake Alaotra, Madagascar. Journal for Nature Conservation, in press. http://doi.org/10.1016/j.jnnfm.2006.12.005

Lanfranconi, .A, Hutton, M., Brugnoli, E. and P. Muniz. 2009. New record of the alien mollusc Rapana venosa (Valenciennes 1846) in the Uruguayan coastal zone of Río de la Plata. Pan-American Journal of Aquatic Sciences, 4: 216-221.

Lanfranconi, A., Brugnoli, E. and P. Muniz. 2013. Preliminary estimates of consumption rates of Rapana venosa (Gastropoda,Muricidae); a new threat to mollusk biodiversity in the Río de la Plata. Aquatic Invasions 8(4): 437-442.

Langbein, L., & Knack, S. (2010). The Worldwide Governance Indicators: Six, One, or None? Journal of Development Studies, 46(2), 350–370. http://doi.org/10.1080/00220380902952399

Langdon, B., Pauchard, A., & Aguayo, M. (2010). Pinus contorta invasion in the Chilean Patagonia: Local patterns in a global context. Biological Invasions, 12(12), 3961–3971. https://doi.org/10.1007/s10530-010-9817-5

Lange, D. 1998. Europe's medicinal and aromatic plants: their use, trade and conservation. Europe's Medicinal and Aromatic Plants: their Use, Trade and Conservation.

Lapola, D.M., Martinelli, L.A., Peres, C.A., Ometto, J.P., Ferreira, M.E., Nobre, C.A., Aguiar, A.P.D., Bustamante, M.M., Cardoso, M.F., Costa, M.H. and Joly, C.A., 2014. Pervasive transition of the Brazilian land-use system. Nature climate change, 4(1), p.27.

Larsen, J.N., O.A. Anisimov, A. Constable, A.B. Hollowed, N. Maynard, P. Prestrud, T.D. Prowse, and J.M.R. Stone, 2014: Polar regions. In: Climate Change 2014: Impacts, Adaptation, and Vulnerability. Part B: Regional Aspects. Contribution of Working Group II to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change [Barros, V.R., C.B. Field, D.J. Dokken, M.D. Mastrandrea, K.J. Mach, T.E. Bilir, M. Chatterjee, K.L. Ebi, Y.O. Estrada, R.C. Genova, B. Girma, E.S. Kissel, A.N. Levy, S. MacCracken, P.R. Mastrandrea, and L.L. White (eds.)]. Cambridge University Press Cambridge, United Kingdom and New York, NY, USA, pp. 1567-1612.

Laufer, G., Gobel, N., Borteiro, C., Soutullo, A., Martínez-Debat, C. & R. O de Sa. 2017. Current status of American bullfrog, Lithobates catesbeianus, invasion in Uruguay and exploration of chytrid infection. Biol Invasions.

Laurance, W. F. (2004). The perils of payoff: corruption as a threat to global biodiversity. Trends in Ecology and Evolution, 19(8), 0–2.

Laurance, W. F., G. R. Clements, S. Sloan, C. S. O'Connell, N. D. Mueller, M. Goosem, O. Venter, D. P. Edwards, B. Phalan, A. Balmford, R. Van Der Ree, and I. B. Arrea. 2014. A global strategy for road building. Nature 513:229–232.

Lauvset, S.K., Gruber, N., Landschützer, P., Olsen, A. & Tjiputra, J., 2015. Trends and drivers in global surface ocean pH over the past 3 decades. Biogeosciences, 12(5), p.1285.

Law, K., 2010. Atmospheric chemistry: More ozone over North America. Nature, 463(7279), pp.307-308.

Le Maitre, D. C., Gaertner, M., Marchante, E., Ens, E. J., Holmes, P. M., Pauchard, A., O'Farrell, P. J., Rogers, A. M., Blanchard, R., Blignaut, J., & Richardson, D. M. (2011). Impacts of invasive Australian acacias: Implications for management and restoration. Diversity and Distributions, 17(5), 1015–1029. https://doi.org/10.1111/j.1472-4642.2011.00816.x

Leach, M. K., and T. J. Givnish. (1996). Ecological determinants of species loss in remnant prairies. Science 273 : 1555 – 1558.

Leadley, P.W., Krug, C.B., Alkemade, R., Pereira, H.M., Sumaila U.R., Walpole, M., Marques, A., Newbold, T., Teh, L.S.L, van Kolck, J., Bellard, C., Januchowski-Hartley, S.R. and Mumby, P.J. (2014). Progress towards the Aichi Biodiversity Targets: An Assessment of Biodiversity Trends, Policy Scenarios and Key Actions. Secretariat of the Convention on Biological Diversity, Montreal, Canada. Technical Series 78, 500 pages

Leatherman, S., Defraene, N. (2006). 10 Most Hurricane Vulnerable Areas. Miami, Florida. <http://www.fema.gov/ emergency/reports/2006/nat112006.shtm.>.

LeBauer, D.S. and Treseder, K.K., 2008. Nitrogen limitation of net primary productivity in terrestrial ecosystems is globally distributed. Ecology, 89(2), pp.371-379.

Lee, P.K., Brook, J.R., Dabek-Zlotorzynska, E. and Mabury, S.A., 2003. Identification of the major sources contributing to PM2. 5 observed in Toronto. Environmental science & technology, 37(21), pp.4831-4840.

Lembrechts, J. J., Milbau, A., & Nijs, I. (2015). Trade-off between competition and facilitation defines gap colonization in mountains. AoB Plants, 1–13. https://doi.org/10.1093/aobpla/plv128

Lemos, M. C., and A. Agrawal. 2006. Environmental governance. Annual Review of Environment and Resources 31:297–325.

Lenton. T. M. 2011. Early warning of climate tipping points. Review article. Nature Climate Change| VOL 1 | JULY 2011 | www.nature.com/natureclimatechange. DOI: 10.1038/NCLIMATE1143

Lenoir, S., Beaugrand, G., & Lecuyer, É. (2011). Modelled spatial distribution of marine fish and projected modifications in the North Atlantic Ocean. Global Change Biology, 17(1), 115–129. https://doi.org/10.1111/j.1365-2486.2010.02229.x

Lepš, J., & Rejmánek, M. (1991). Convergence or divergence: what should we expect from vegetation succession?. Oikos, 261-264.

Lesica P (2014) Arctic-alpine plants decline over two decades in Glacier National Park, Montana, U.S.A. Arctic Antarctic and Alpine Research, 46, 327-332.

Lewis, J., Hoover, J., & MacKenzie, D. (2017). Mining and Environmental Health Disparities in Native American Communities. Current Environmental Health Reports, 4, 130-141, 10.1007/s40572-017-0140-5

Lewison, R. L., S. A. Freeman, and L. B. Crowder. 2004. Quantifying the effects of fisheries on threatened species-loggerhead & leatherback turtles. Ecology Letters 7:221–231.

Li, C., Balluz, L.S., Vaidyanathan, A., Wen, X.J., Hao, Y. and Qualters, J.R., 2016. Long-Term Exposure to Ozone and Life Expectancy in the United States, 2002 to 2008. Medicine, 95(7).

Liebezeit, J.R., S.J. Kendall, S. Brown, C.B. Johnson, P. Martin, T.L. McDonald, D.C. Payer, C.L. Rea, B. Streever, A.M. Wildman, and S. Zack. 2009. Influence of human development and predators on nest survival of tundra birds, Arctic Coastal Plain, Alaska. Ecological Applications 19:1628-1644.

Lima, L. S., Coe, M. T., Soares Filho, B. S., Cuadra, S. V., Dias, L. C. P., Costa, M. H., Lima, L. S., & Rodrigues, H. O. (2014). Feedbacks between deforestation, climate, and hydrology in the Southwestern Amazon: Implications for the provision of ecosystem services. Landscape Ecology, 29(2), 261–274. https://doi.org/10.1007/s10980-013-9962-1

Lindenmayer, D. B., & Likens, G. E. (2009). Adaptive monitoring: a new paradigm for long-term research and monitoring. Trends in Ecology & Evolution, 24(9), 482-486.

Liu, J., Hull, V., Batistella, M., DeFries, R., Dietz, T., Fu, F., Hertel, T. W., Izaurralde, R. C., Lambin, E. F., Li, S., Martinelli, L. A., McConnell, W. J., Moran, E. F., Naylor, R., Ouyang, Z., Polenske, K. R., Reenberg, A., de Miranda Rocha, G., Simmons, C. S., Verburg, P. H., Vitousek, P. M., Zhang, F., & Zhu, C. (2013). Framing Sustainability in a Telecoupled World. Ecology and Society, 18(2), art26. https://doi.org/10.5751/ES-05873-180226

Liu, J., You, L., Amini, M., Obersteiner, M., Herrero, M., Zehnder, A. J., & Yang, H. (2010). A high-resolution assessment on global nitrogen flows in cropland. Proceedings of the National Academy of Sciences, 107(17), 8035-8040.

Liu, Y., S.-K. Lee, D. B. Enfield, B. A. Muhling, J. T. Lamkin, F. E. Muller-Karger, and M. A. Roffer, 2015: Potential impact of climate change on the Intra-Americas Seas: Part-1. A dynamic downscaling of the CMIP5 model projections. J. Marine Syst., 148, 56-69, doi:10.1016/j.jmarsys.2015.01.007.

Liu, Y., S.-K. Lee, D. B. Enfield, B. A. Muhling, J. T. Lamkin, F. E. Muller-Karger, and M. A. Roffer, 2016: Past and future climate variability in the Intra-Americas Sea and its impact on the marine ecosystem and fisheries, US CLIVAR Variations, Vol. 14, No. 1, 27-32.

Liverman, D. M., & Vilas, S. (2006). Neoliberalism and the Environment in Latin America. Annual Review of Environment and Resources, 31(1), 327–363. http://doi.org/10.1146/annurev.energy.29.102403.140729

Lôbo, D., Leão, T., Melo, F. P. L., Santos, A. M. M., & Tabarelli, M. (2011). Forest fragmentation drives Atlantic forest of northeastern Brazil to biotic homogenization. Diversity and Distributions, 17(2), 287–296. https://doi.org/10.1111/j.1472-4642.2010.00739.x

Loh, T.-L., S. E. McMurray, T. P. Henkel, J. Vicente, and J. R. Pawlik. 2015. Indirect effects of overfishing on Caribbean reefs: sponges overgrow reef-building corals. PeerJ 3:e901. <http://www.pubmedcentral.nih.gov/articlerender.fcgi?artid=4419544&tool=pmcentrez&rendertype=abstract>.

Lopes, A., and M. T. F. Piedade. 2014. Experimental study on the survival of the water hyacinth (Eichhornia crassipes (Mart.) Solms--Pontederiaceae) under different oil doses and times of exposure. Environmental science and pollution research international 21:13503–11.

Lopes, R. M., Coradin, L., Beck, V., & Rimoldi, D. (2009). Informe sobre as Espécies Exóticas Invasoras Marinhas no Brasil. Brasília, Ministério do Meio Ambiente, Série Biodiversidade (Vol. 33).

Lopez-Lanus, B., I. Roesler, D. E. Blanco, P. F. Petracci, M. Serra, and M. E. Zaccagnini. 2007. Bobolink (dolichonyx oryzivorus) numbers and non breeding ecology in the rice fields of San Javier, Santa Fe province, Argentina. Ornitologia Neotropical 18:493–502. <<Go to ISI>://000252231000002>.

Lopez, C.B., Jewett, E.B., Dortch, Q., Walton, B.T., Hundell, H. K. 2008. Scientific Assessment of Marine Harmful Algal Blooms. Interagency Working Group on Harmful Algal Blooms, Hypoxia, and Human Health of the Joint Sucommittee on Ocean Science and Technology.

Lovingood, T., Parker, B., Smith, T. N., Canes, H., Fennell, F., Cofer, D., & Reilly, T. (2004). Nationwide identification of hardrock mining sites. Office of Inspector General (OIG) of the US Environmental Protection Agency, Washington, DC, 〈 http://www. epa. gov/oig/reports/2004/20040331-2004-p-00005. pdf〉 (Nov. 1, 2012).

Lozano, P., Bussmann, R. W., & Küppers, M. (2006). Landslides as ecosystem disturbance-their implications and importance in Southern Ecuador. Lyonia, 9, 75-81.

Lugo, A. E. 1998. Mangrove forests: A tough system to invade but an easy one to rehabilitate. Marine Pollution Bulletin 37:427–430.

Luiz, O. J., Floeter, S. R., Rocha, L. A., & Ferreira, C. E. L. (2013). Perspectives for the lionfish invasion in the South Atlantic: Are Brazilian reefs protected by the currents? Marine Ecology Progress Series, 485, 1–7.

Lutz, D. A., R. L. Powell, and M. R. Silman. 2013. Four Decades of Andean Timberline Migration and Implications for Biodiversity Loss with Climate Change. PLoS ONE 8:e74496.

Lyon, J. S., T. J. Hilliard, and T. N. Bethell. 1993. Burden of Gilt: The legacy of environmental damage from abandoned mines, and what America should do about it. Mineral Policy Center.

Lyra-Neves, R. M. De, Oliveira, M. a. B., Telino-Júnior, W. R., & Santos, E. M. Dos. (2007). Comportamentos interespecíficos entre Callithrix jacchus (Linnaeus) (Primates, Callitrichidae) e algumas aves de Mata Atlântica, Pernambuco, Brasil. Revista Brasileira de Zoologia, 24(3), 709–716.

Macdonald, R.W., Barrie, L.A., Bidleman, T.F., Diamond, M.L., Gregor, D.J., Semkin, R.G., Strachan, W.M.J., Li, Y.F., Wania, F., Alaee, M. and Alexeeva, L.B., 2000. Contaminants in the Canadian Arctic: 5 years of progress in understanding sources, occurrence and pathways. Science of the Total Environment, 254(2), pp.93-234.

MacDougall, A. S., & Turkington, R. (2005). Are invasive species the drivers or passengers of change in degraded ecosystems? Ecology, 86(1), 42–55.

Machado, W., and L. D. Lacerda. 2004. Overview of the biogeochemical controls and concerns with trace metal accumulation in mangrove sediments. Pages 319-334 in. L. D. Lacerda, R. E. Santelli, E. Duursma, and J. J. Abrao, editors. Environmental geochemistry in tropical and subtropical environments.

Mack, R. N., & Lonsdale, W. M. (2001). Humans as Global Plant Dispersers: Getting More Than We Bargained For: Current introductions of species for aesthetic purposes present the largest single challenge for predicting which plant immigrants will become future pests. BioScience, 51(2), 95-102.

Mack, R. N., & Thompson, J. N. (1982). Evolution in steppe with few large, hooved mammals. The American Naturalist, 119(6), 757-773.

Mack, R., Simberloff, D., Lonsdale, W., Evans, H., Clout, M., & Bazzaz, F. (2000). Biotic Invasions: Causes, Epidemiology, Global Consequences, and Control. Ecological Applications, 10(3), 689–710.

Madrigal-González, J., Cea, A. P., Sánchez-Fernández, L. A., Martínez-Tillería, K. P., Calderón, J. E., & Gutiérrez, J. R. (2013). Facilitation of the non-native annual plant Mesembryanthemum crystallinum (Aizoaceae) by the endemic cactus Eulychnia acida (Cactaceae) in the Atacama Desert. Biological Invasions, 15(7), 1439–1447. https://doi.org/10.1007/s10530-012-0382-y

Maffi, L. (2005). Linguistic, cultural, and biological diversity. Annu. Rev. Anthropol., 34, 599-617.

Magalhães, J. L. L., M. A. Lopes, and H. L. De Queiroz. 2015. Development of a Flooded Forest Anthropization Index (FFAI) applied to Amazonian areas under pressure from different human activities. Ecological Indicators 48:440–447.

Magrin, G.O., J.A. Marengo, J.-P. Boulanger, M.S. Buckeridge, E. Castellanos, G. Poveda, F.R. Scarano, and S. Vicuña, 2014: Central and South America. In: Climate Change 2014: Impacts, Adaptation, and Vulnerability. Part B: Regional Aspects. Contribution of Working Group II to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change [Barros, V.R., C.B. Field, D.J. Dokken, M.D. Mastrandrea, K.J. Mach, T.E. Bilir, M. Chatterjee, K.L. Ebi, Y.O. Estrada, R.C. Genova, B. Girma, E.S. Kissel, A.N. Levy, S. MacCracken, P.R. Mastrandrea, and L.L. White (eds.)]. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA, pp. 1499-1566.

Magris, R. A., & Barreto, R. (2010). Mapping and assessment of protection of mangrove habitats in Brazil. Pan-American Journal of Aquatic Sciences, 5(4), 546-56.

Mahaffey, K.R. and Mergler, D., 1998. Blood levels of total and organic mercury in residents of the upper St. Lawrence River basin, Quebec: association with age, gender, and fish consumption. Environmental research, 77(2), pp.104-114.

Maki, A.W. 1992. Annual review of measured risks: The environmental impacts of the Prudhoe Bay, Alaska, oil field. Environmental Toxicology and Chemistry 11:1691-1707.

Malcom, J.R., Liu, C., Neilson, R.P., Hansen, L., Hannah, L. 2005. Global warming and extinctions of endemic species from biodiversity hotspots. Conservation Biology Volume 20, No. 2, 538–548

Malhi, Y., and J. Wright. 2004. Spatial patterns and recent trends in the climate of tropical rainforest regions. Philosophical transactions of the Royal Society of London. Series B, Biological sciences 359:311–29.

Malm, O. 1998. Gold mining as a source of mercury exposure in the Brazilian Amazon. Environmental research 77:73–8.

Mann R, Occhipinti A & Harding JM (eds.). 2004. Alien Species Alert: Rapana venosa (veined whelk). ICES Cooperative Research Report, no 264, 14 pp.

Mantyka-Pringle, C. S., Martin, T. G., & Rhodes, J. R. (2012). Interactions between climate and habitat loss effects on biodiversity: a systematic review and meta-analysis. Global Change Biology, 18(4), 1239-1252.

Manuschevich, D. (2016). Neoliberalization of forestry discourses in Chile. Forest Policy and Economics, 69, 21–30. http://doi.org/10.1016/j.forpol.2016.03.006

Manzello, D.P., 2010. Ocean acidification hotspots: spatiotemporal dynamics of the seawater CO2 system of eastern Pacific coral reefs. Limnology and Oceanography, 55(1), pp.239-248.

Marchese, C. (2015). Biodiversity hotspots: A shortcut for a more complicated concept. Global Ecology and Conservation, 3, 297-309.

Marengo, J. A. 2004. Interdecadal variability and trends of rainfall across the Amazon basin. Theoretical and Applied Climatology 78. <http://link.springer.com/10.1007/s00704-004-0045-8>. Accessed 6 Jun 2016.

Marengo, J. A., & Espinoza, J. C. (2016). Extreme seasonal droughts and floods in Amazonia: causes, trends and impacts. International Journal of Climatology, 36(3), 1033-1050.

Marengo, J. A., R. Jones, L. M. Alves, and M. C. Valverde. 2009. Future change of temperature and precipitation extremes in south america as derived from the precis regional climate modeling system. International Journal of Climatology 29:2241–2255.

Marengo, J. A., J. D. Pabón, A. Díaz, G. Rosas, E. Montealegre, M. Villacis, and S. Solman. 2011. Climate Change : Evidence and Future Scenarios for the Andean Region. Pages 110–127 in S. K. Herzog, R. Martinez, P. M. Jørgensen, and H. Tiessen, editors. Climate Change an Biodiversity in the Tropical

Andes. Inter-American Institute for Global Change Research (IAI) and Scientific Committee on Problems of the Environment (SCOPE).

Marengo, J. A., S. C. Chou, G. Kay, L. M. Alves, J. F. Pesquero, W. R. Soares, D. C. Santos, A. A. Lyra, G. Sueiro, R. Betts, D. J. Chagas, J. L. Gomes, J. F. Bustamante, and P. Tavares. 2012. Development of regional future climate change scenarios in South America using the Eta CPTEC/HadCM3 climate change projections: climatology and regional analyses for the Amazon, São Francisco and the Paraná River basins. Climate Dynamics 38:1829–1848. Springer-Verlag.

Marengo, J. A., Nunes, L. H., Souza, C. R. G., Harari, J., Muller-Karger, F., Greco, R., Hosokawa, E. K., Tabuchi, E. K., Merrill, S. B., Reynolds, C. J., Pelling, M., Alves, L. M., Aragão, L. E., Chou, S. C., Moreira, F., Paterson, S., Lockman, J. T., & Gray, A. G. (2017). A globally deployable strategy for co-development of adaptation preferences to sea-level rise: the public participation case of Santos, Brazil. Natural Hazards, 88(1), 39–53. https://doi.org/10.1007/s11069-017-2855-x

Maret T.R. & MacCoy D.E. (2002). Fish Assemblages and Environmental Variables Associated with Hard Rock Mining in the Coeur d'Alene River Basin, Idaho. Transactions of the American Fisheries Society, 131, 865-884, doi:10.1577/1548-8659(2002)131<0865:FAAEVA>2.0.CO;2

Maret, T. R., Cain, D. J., MacCoy, D. E., & Short, T.M. (2003). Response of benthic invertebrate assemblages to metal exposure and bioaccumulation associated with hard-rock mining in northwestern streams, USA. Journal of the North American Benthological Society, 22, 598-620, 10.2307/1468356

Marshall, V. M., Lewis, M. M., & Ostendorf, B. (2012). Buffel grass (Cenchrus ciliaris) as an invader and threat to biodiversity in arid environments: A review. Journal of Arid Environments, 78, 1–12. https://doi.org/10.1016/j.jaridenv.2011.11.005

Martin, L. M., Moloney, K. A., & Wilsey, B. J. (2005). An assessment of grassland restoration success using species diversity components. Journal of Applied Ecology, 42(2), 327-336.

Martin, P. H., Canham, C. D., & Marks, P. L. (2009). Why forests appear resistant to exotic plant invasions: Intentional introductions, stand dynamics, and the role of shade tolerance. Frontiers in Ecology and the Environment, 7(3), 142–149. https://doi.org/10.1890/070096

Martínez AS, Masciocchi M, Villacide JM, Huerta G, Daneri L, Bruchhausen A, Rozas G, Corley JC (2013) Ashes in the air: the effects of volcanic ash emissions on plant–pollinator relationships and possible consequences for apiculture. Apidologie 44: 268–277

Martínez, M.O., Napolitano, D.A., MacLennan, G.J., O'Callaghan, C., Ciborowski, S. and Fabregas, X., 2007. Impacts of petroleum activities for the Achuar people of the Peruvian Amazon: summary of existing evidence and research gaps. Environmental Research Letters, 2(4), p.045006.

Martínez-Alier, J. (2014). The environmentalism of the poor. Geoforum, 54, 239–241. http://doi.org/10.1016/j.geoforum.2013.04.019

Martínez, V., & Castillo, O. L. (2016). The political ecology of hydropower: Social justice and conflict in Colombian hydroelectricity development. Energy Research and Social Science, 22, 69–78.

Martínez-López, E., Espín, S., Barbar, F., Lambertucci, S.A., Gómez-Ramírez, P. and García-Fernández, A.J., 2015. Contaminants in the southern tip of South America: Analysis of organochlorine compounds in feathers of avian scavengers from Argentinean Patagonia. Ecotoxicology and environmental safety, 115, pp.83-92.

Martins, C. R. (2006). Caracterização e manejo da gramínea Melinis minutiflora P. Beauv. (capim-gordura): uma espécie invasora do Cerrado.

Martins, C.C., Bícego, M.C., Mahiques, M.M., Figueira, R.C., Tessler, M.G. and Montone, R.C., 2011. Polycyclic aromatic hydrocarbons (PAHs) in a large South American industrial coastal area (Santos Estuary, Southeastern Brazil): sources and depositional history. Marine pollution bulletin, 63(5), pp.452-458.

Marubini, F. and Davies, P.S., 1996. Nitrate increases zooxanthellae population density and reduces skeletogenesis in corals. Marine Biology, 127(2), pp.319-328.

Mascia, M. B., Pailler, S., Krithivasan, R., Roshchanka, V., Burns, D., Mlotha, M. J., Murray, D. R., & Peng, N. (2014). Protected area downgrading, downsizing, and degazettement (PADDD) in Africa, Asia, and Latin America and the Caribbean, 1900-2010. Biological Conservation, 169, 355–361. https://doi.org/10.1016/j.biocon.2013.11.021

Masek, J. G., Cohen, W. B., Leckie, D., Wulder, M. A., Vargas, R., de Jong, B., Healey, S., Law, B., Birdsey, R., Houghton, R. A., Mildrexler, D., Goward, S., & Smith, W. B. (2011). Recent rates of forest harvest and conversion in North America. Journal of Geophysical Research, 116(G4), G00K03. https://doi.org/10.1029/2010JG001471

Masi, E., Pino, F. A., Santos, M. das G. S., Genehr, L., Albuquerque, J. O. M., Bancher, A. M., & Alves, J. C. M. (2010). Socioeconomic and environmental risk factors for urban rodent infestation in Sao Paulo, Brazil. Journal of Pest Science, 83(3), 231–241. https://doi.org/10.1007/s10340-010-0290-9

Mathis, J.T., Cross, J.N., Evans, W. and Doney, S.C., 2015. Ocean acidification in the surface waters of the Pacific-Arctic boundary regions. Oceanography, 28(2), pp.122-135.

Maxim, L., Spangenberg, J. H., & O'Connor, M. (2009). An analysis of risks for biodiversity under the DPSIR framework. Ecological Economics, 69(1), 12-23.

McClintock, J.B., Angus, R.A., Mcdonald, M.R., Amsler, C.D., Catledge, S.A. and Vohra, Y.K., 2009. Rapid dissolution of shells of weakly calcified Antarctic benthic macroorganisms indicates high vulnerability to ocean acidification. Antarctic Science, 21(5), pp.449-456.

McConnell, J.R. and Edwards, R., 2008. Coal burning leaves toxic heavy metal legacy in the Arctic. Proceedings of the National Academy of Sciences, 105(34), pp.12140-12144.

McDonald, R.I., Weber, K., Padowski, J., Flörke, M., Schneider, C., Green, P.A., Gleeson, T., Eckman, S., Lehner, B., Balk, D. and Boucher, T., 2014. Water on an urban planet: Urbanization and the reach of urban water infrastructure. Global Environmental Change, 27, pp.96-105.

McEwan, R. W., Dyer, J. M., & Pederson, N. (2011). Multiple interacting ecosystem drivers: toward an encompassing hypothesis of oak forest dynamics across eastern North America. Ecography, 34(2), 244-256.

McGraw, J. B., S. Souther, and A. E. Lubbers. 2010. Rates of Harvest and Compliance with Regulations in Natural Populations of American Ginseng (Panax quinquefolius L.). Natural Areas Journal 30:202–210.

Mcguire, A. D., F. S. C. Iii, J. E. Walsh, and C. Wirth. 2006. Integrated regional changes in Arctic climate f eedbacks : Implications for the Global. 61–91.

McLinden, C.A., Fioletov, V., Krotkov, N.A., Li, C., Boersma, K.F. and Adams, C., 2015. A decade of change in NO2 and SO2 over the Canadian oil sands as seen from space. Environmental science & technology, 50(1), pp.331-337.

McNeil, B.I., & Matear, R.J. (2008). Southern Ocean acidification: A tipping point at 450-ppm atmospheric CO₂. Proceedings of the National Academy of Sciences, 105, 18860-18864, 10.1073/pnas.0806318105

McNeish, John-Andrew. "Extraction, protest and indigeneity in Bolivia: the TIPNIS effect." Latin American and Caribbean Ethnic Studies 8.2 (2013): 221-242.

McPhearson T., Auch R., Alberti M. (2013) Regional Assessment of North America: Urbanization Trends, Biodiversity Patterns, and Ecosystem Services. In: Elmqvist T. et al. (eds) Urbanization, Biodiversity and Ecosystem Services: Challenges and Opportunities. Springer, Dordrecht

Meissner, K. J., Lippmann, T., & Gupta, A. S. (2012). Large-scale stress factors affecting coral reefs: open ocean sea surface temperature and surface seawater aragonite saturation over the next 400 years. Coral Reefs, 31(2), 309-319.

Mekonnen, M. M., M. Pahlow, M. M. Aldaya, E. Zarate, and A. Y. Hoekstra. 2015. Sustainability, efficiency and equitability of water consumption and pollution in latin America and the Caribbean. Sustainability (Switzerland) 7:2086–2112.

Melillo, J. M., T. (T. C. . Richmond, and G. W. Yohe. 2014. Climate Change Impacts in the United States.

Mendelsohn, R., Emanuel, K., Chonabayashi, S., & Bakkensen, L. (2012). The impact of climate change on global tropical cyclone damage. Nature climate change, 2(3), 205-209.

Mendes, D., and J. A. Marengo. 2010. Temporal downscaling: A comparison between artificial neural network and autocorrelation techniques over the Amazon Basin in present and future climate change scenarios. Theoretical and Applied Climatology 100:413–421.

Méndez-Lázaro, P., Muller-Karger, F.E., Otis, D., McCarthy, Matthew J., and Rodríguez, Ernesto. 2017. A heat vulnerability index to improve urban public health management in San Juan, Puerto Rico. International Journal of Biometeorology. doi:10.1007/s00484-017-1319-z.

Méndez-Lázaro, P., Nieves-Santiango, A., & Miranda-Bermúdez, J. (2014). Trends in total rainfall, heavy rain events, and number of dry days in San Juan, Puerto Rico, 1955-2009. Ecology and Society, 19(2).
Mendonça, A. F., T. Armond, A. C. L. Camargo, N. F. Camargo, J. F. Ribeiro, P. L. Zangrandi, and E. M. Vieira. 2015. Effects of an extensive fire on arboreal small mammal populations in a neotropical savanna woodland. Journal of Mammalogy 96:368–379.
Merritt, D. M., & Leroy Poff, N. (2010). Shifting dominance of riparian Populus and Tamarix along gradients of flow alteration in western North American rivers. Ecological Applications, 20(1), 135–152. https://doi.org/10.1890/08-2251.1
Meserve, P. L., Kelt, D. A., Gutiérrez, J. R., Previtali, M. A., & Milstead, W. B. (2016). Biotic interactions and community dynamics in the semiarid thorn scrub of Bosque Fray Jorge National Park, north-central Chile: A paradigm revisited. Journal of Arid Environments, 126, 81–88. https://doi.org/10.1016/j.jaridenv.2015.08.016
Meyfroidt, P., Carlson, K. M., Fagan, M. E., Gutiérrez-Vélez, V. H., Macedo, M. N., Curran, L. M., DeFries, R. S., Dyer, G. A., Gibbs, H. K., Lambin, E. F., Morton, D. C., & Robiglio, V. (2014). Multiple pathways of commodity crop expansion in tropical forest landscapes. Environmental Research Letters, 9(7), 74012. https://doi.org/10.1088/1748-9326/9/7/074012
Michalak, A.M., Anderson, E.J., Beletsky, D., Boland, S., Bosch, N.S., Bridgeman, T.B., Chaffin, J.D., Cho, K., Confesor, R., Daloğlu, I. and DePinto, J.V., 2013. Record-setting algal bloom in Lake Erie caused by agricultural and meteorological trends consistent with expected future conditions. Proceedings of the National Academy of Sciences, 110(16), pp.6448-6452.
Michener, W. K., E. R. Blood, K. L. Bildstein, M. M. Brinson, and L. R. Gardner. 1997. Climate Change , Hurricanes and Tropical Storms , and Rising Sea Level in Coastal Wetlands. Ecological Applications 7:770–801.
Middeldorp, N., Morales, C., & van der Haar, G. (2016). Social mobilisation and violence at the mining frontier: The case of Honduras. Extractive Industries and Society, 3(4), 930–938. http://doi.org/10.1016/j.exis.2016.10.008
Miglioranza, K.S., Gonzalez, M., Ondarza, P.M., Shimabukuro, V.M., Isla, F.I., Fillmann, G., Aizpún, J.E. and Moreno, V.J., 2013. Assessment of Argentinean Patagonia pollution: PBDEs, OCPs and PCBs in different matrices from the Río Negro basin. Science of the Total Environment, 452, pp.275-285.
Millar, C. I., & Stephenson, N. L. (2015). Temperate forest health in an era of emerging megadisturbance. Science, 349(6250), 823-826.
Millennium Ecosystem Assessment,(MEA) 2005. Ecosystems and Human Well-being: Synthesis. Island Press, Washington, DC. Copyright © 2005 World Resources Institute 139 pp
Miller, K. A., Aguilar-Rosas, L. E., & Pedroche, F. F. (2011). A review of non-native seaweeds from California, USA and Baja California, Mexico = Reseña de algas marinas no nativas de California, EUA y Baja California, México. Hidrobiológica, 21(3), 365–379. Retrieved from http://www.vliz.be/en/imis?module=ref&refid=218235
Miller, M. J. (2011). Persistent Illegal Logging in Costa Rica: The Role of Corruption Among Forestry Regulators. http://doi.org/10.1177/1070496510394319
Mills, D.J., Westlund, L., de Graaf, G., Kura, Y., Willman, R. and Kelleher, K., 2011. Under-reported and undervalued: small-scale fisheries in the developing world. Small-scale fisheries management: frameworks and approaches for the developing world, pp.1-15.
Miloslavich, P., Martín, A., Klein, E., Díaz, Y., Lasso, C. A., Cárdenas, J. J., & Lasso-Alcalá, O. M. (2011). Biodiversity and Conservation of the Estuarine and Marine Ecosystems of the Venezuelan Orinoco Delta. In Ecosystems Biodiversity. InTech.
Mistry, J., A. Berardi, V. Andrade, T. Kraho, P. Krah ˆ o, and O. Leonardos. ˆ 2005. Indigenous fire management in the cerrado of Brazil: the case of the Kraho of Tocant ˆ ´ıns. Human Ecology 33:365–386.
Mitchell, R.T., Blagbrough, H.P. and VanEtten, R.C., 1953. The effects of DDT upon the survival and growth of nestling songbirds. The Journal of Wildlife Management, 17(1), pp.45-54.
Mitsch, W. J., & Hernandez, M. E. (2013). Landscape and climate change threats to wetlands of North and Central America. Aquatic sciences, 75(1), 133-149.

Modernel, P., Rossing, W. A., Corbeels, M., Dogliotti, S., Picasso, V., & Tittonell, P. (2016). Land use change and ecosystem service provision in Pampas and Campos grasslands of southern South America. Environmental Research Letters, 11(11), 113002.

Mol, J. H., J. S. Ramlal, C. Lietar, and M. Verloo. 2001. Mercury Contamination in Freshwater, Estuarine, and Marine Fishes in Relation to Small-Scale Gold Mining in Suriname, South America. Environmental Research Section A 86:183–197. <http://www.idealibrary.com>.

Molinos-Senante, M., & Sala-Garrido, R. (2015). The impact of privatization approaches on the productivity growth of the water industry: A case study of Chile. Environmental Science & Policy, 50, 166–179. http://doi.org/http://dx.doi.org/10.1016/j.envsci.2015.02.015

Moller, H., Berkes, F., Lyver, P. O. B., & Kislalioglu, M. (2004). Combining science and traditional ecological knowledge: monitoring populations for co-management. Ecology and society, 9(3).

Morales-Hidalgo, D., S. N. Oswalt, and E. Somanathan. 2015. Status and trends in global primary forest, protected areas, and areas designated for conservation of biodiversity from the Global Forest Resources Assessment 2015. Forest Ecology and Management 352:68–77.

Morang, A., J.D. Rosati, and D.B. King, 2013. Regional Sediment Processes, Sediment Supply, and their Impact on the Louisiana Coast. Journal of Coastal Research 63, 141-165.

Morell, V. 2017. World's most endangered marine mammal down to 30. Science (Wash.) 355(6325): 558-559. hdl.handle.net/10.1126/science.355.6325.558

Moreno, T. A., & Huber-Sannwald, E. (2011). Impacts of drought on agriculture in Northern Mexico. In Coping with Global Environmental Change, Disasters and Security (pp. 875-891). Springer, Berlin, Heidelberg.

Moreno-Mateos, D., Barbier, E. B., Jones, P. C., Jones, H. P., Aronson, J., López-López, J. A., McCrackin, M. L., Meli, P., Montoya, D., & Rey Benayas, J. M. (2017). Anthropogenic ecosystem disturbance and the recovery debt. Nature Communications, 8, 14163. https://doi.org/10.1038/ncomms14163

Morris, J.A.J. and J.L. Akins JL. 2009. Feeding ecology of invasive lionfish (Pterois volitans) in the Bahamian Archipelago. Environmental Biology of Fishes 86: 389–398.

Morton, D. C., Y. Le Page, R. DeFries, J. G. Collatz, and G. C. Hurtt. 2013. Understorey fire frequency and the fate of burned forest in southern Amazonia. Philosophical Transactions

Moss, C. 2008. Patagonia. A cultural history. Signal Books. Oxford. 299 pp.

Mueller, N. D., J. S. Gerber, M. Johnston, D. K. Ray, N. Ramankutty, and J. a. Foley. 2012. Closing yield gaps through nutrient and water management. Nature 490:254–257. <http://dx.doi.org/10.1038/nature11420>.

Muhling, B. A., Lamkin, J. T., Alemany, F., García, A., Farley, J., Ingram, G. W., Berastegui, D. A., Reglero, P., & Carrion, R. L. (2017). Reproduction and larval biology in tunas, and the importance of restricted area spawning grounds. Reviews in Fish Biology and Fisheries, 27(4), 697–732. https://doi.org/10.1007/s11160-017-9471-4

Muhling, B. A., Liu, Y., Lee, S. K., Lamkin, J. T., Roffer, M. A., Muller-Karger, F., & Walter, J. F. (2015). Potential impact of climate change on the Intra-Americas Sea: Part 2. Implications for Atlantic bluefin tuna and skipjack tuna adult and larval habitats. Journal of Marine Systems, 148, 1–13. https://doi.org/10.1016/j.jmarsys.2015.01.010

Mueller, S. A., Anderson, J. E., & Wallington, T. J. (2011). Impact of biofuel production and other supply and demand factors on food price increases in 2008. Biomass and Bioenergy, 35(5), 1623-1632.

Muller-Karger, F., Digna Rueda-Roa, Francisco Chavez, Maria Kavanaugh, and Mitchell A. Roffer. 2017. Megaregions Among the Large Marine Ecosystems of the Americas. Environmental Development. http://dx.doi.org/10.1016/j.envdev.2017.01.005

Muller-Karger, Frank E., Joseph P. Smith, Sandra Werner, Robert Chen, Mitchell Roffer, Yanyun Liu, Barbara Muhling, David Lindo-Atichati, John Lamkin, Sergio Cerdeira-Estrada, and David B. Enfield. 2015. Natural Variability of Surface Oceanographic Conditions in the Offshore Gulf of Mexico. Progress in Oceanography. 10.1016/j.pocean.2014.12.007.

Munden, J 2013, Reducing Negative Ecological Impacts of Capture Fisheries Through Gear Modification, Masters Thesis Biology Department, Faculty of Science, Memorial University of Newfoundland, St. Johns, Newfoundland and Labrador

Muñoz, A. A., Celedon-Neghme, C., Cavieres, L. A., & Arroyo, M. T. K. (2005). Bottom-up effects of nutrient availability on flower production, pollinator visitation, and seed output in a high-Andean shrub. Oecologia, 143(1), 126–135. https://doi.org/10.1007/s00442-004-1780-3

Munson, S. M., Belnap, J., & Okin, G. S. (2011). Responses of wind erosion to climate-induced vegetation changes on the Colorado Plateau. Proceedings of the National Academy of Sciences, 108(10), 3854-3859.

Munthe, J., R. a D. Bodaly, B. a Branfireun, C. T. Driscoll, C. C. Gilmour, R. Harris, M. Horvat, M. Lucotte, and O. Malm. 2007. Recovery of mercury-contaminated fisheries. Ambio 36:33–44.

Murcia, C. & Guariguata, M.R. (2014). La Restauración Ecológica en Colombia: Tendencias, necesidades y oportunidades. Documentos Ocasionales 107, Bogor, Indonesia: CIFOR, 98pp.

Murphy, G. E., & Romanuk, T. N. (2014). A meta‐analysis of declines in local species richness from human disturbances. Ecology and evolution, 4(1), 91-103.

Myers-smith, I. H., B. C. Forbes, M. Wilmking, M. Hallinger, T. Lantz, D. Blok, K. D. Tape, M. Macias-fauria, U. Sass-klaassen, and L. Esther. 2011. Shrub expansion in tundra ecosystems : dynamics , impacts and research priorities. 045509.

Myers, N., and J. Kent (2003). "New consumers: the influence of affluence on environment".Published online 2 April 2003: https://www.ncbi.nlm.nih.gov/pmc/articles/PMC153663/

Myers, N., and J. Kent (2003). "New consumers: the influence of affluence on environment". Published online 2 April 2003: https://www.ncbi.nlm.nih.gov/pmc/articles/PMC153663/

Nascimento Jr., W. R., P. W. M. Souza-Filho, C. Proisy, R. M. Lucas, and A. Rosenqvist. 2013. Mapping changes in the largest continuous Amazonian mangrove belt using object-based classification of multisensor satellite imagery. Estuarine, Coastal and Shelf Science 117:83-93.

Nasi, R., A. Taber, and N. van Vliet. 2011. Empty forests, empty stomachs? Bushmeat and livelihoods in the Congo and Amazon Basins. International Forestry Review 13:355–368.

Nasi, R., D. Brown, D. Wilkie, E. Bennett, C. Tutin, G. Van Tol, and T. Christophersen. 2008. Conservation and use of wildlife-based resources: the bushmeat crisis. Diversity. Volume 33. <http://www.cbd.int/doc/publications/cbd-ts-33-en.pdf>.

National Oceanic and Atmospheric Administration. Global impacts of El Niño and La Niña. <https://www.climate.gov/news-features/featured-images/global-impacts-el-ni%C3%B1o-and-la-ni%C3%B1a>. Accessed 6 Jun 2013.

Nations, F. A. O. (2013). Current world fertilizer trends and outlook to 2015; Rome.

Nations, F. A. O. (2017). World fertilizer trends and outlook to 2020. SUMMARY REPORT

United Nations & The World Bank, (2010). Natural hazards, unnatural disasters: the economics of effective prevention. The World Bank.

Neff, J. C., A. P. Ballantyne, G. L. Farmer, N. M. Mahowald, J. L. Conroy, C. C. Landry, J. T. Overpeck, T. H. Painter, C. R. Lawrence, and R. L. Reynolds. 2008. Increasing eolian dust deposition in the western United States linked to human activity. Nature Geoscience 1:189–195.

Nellemann, C., & Corcoran, E. (2010). Dead planet, living planet: biodiversity and ecosystem restoration for sustainable development. United Nations Environment Programme (UNEP).

Nelson, E., G. Mendoza, J. Regetz, S. Polasky, H. Tallis, D. R. Cameron, K. M. A. Chan, G. C. Daily, J. Goldstein, P. M. Kareiva, E. Lonsdorf, R. Naidoo, T. H. Ricketts, and M. R. Shaw. 2009. Modeling multiple ecosystem services, biodiversity conservation, commodity production, and tradeoffs at landscape scales. Frontiers in Ecology and the Environment 7:4–11.

Nelson, G. C., Dobermann, A., Nakicenovic, N., & O'Neill, B. C. (2006). Anthropogenic drivers of ecosystem change: an overview. Ecology and Society, 11(2).

Neme, L. (2016). In Latin America, Illegal Wildlife Trade Explodes, Huffington Post, http://www.huffingtonpost.com/laurel-neme/in-latin-america-illegal-_b_8492020.html

Nepstad, D., Schwartzman, S., Bamberger, B., Santilli, M., Ray, D., Schlesinger, P., Lefebvre, P., Alencar, A., Prinz, E., Fiske, G., & Rolla, A. (2006). Inhibition of Amazon deforestation and fire by parks and indigenous lands. Conservation Biology, 20(1), 65–73. https://doi.org/10.1111/j.1523-1739.2006.00351.x

Newbold, T., L. N. Hudson, S. L. Hill, S. Contu, I. Lysenko, R. A. Senior, L. Börger, D. J. Bennett, A. Choimes, B. Collen, J. Day, A. De Palma, S. Dıáz, S. Echeverria-Londoño, M. J. Edgar, A. Feldman, M. Garon, M.

L. K. Harrison, T. Alhusseini, D. J. Ingram, Y. Itescu, J. Kattge, V. Kemp, L. Kirkpatrick, M. Kleyer, D. Laginha Pinto Correia, C. D. Martin, S. Meiri, M. Novosolov, Y. Pan, H. R. P. Phillips, D. W. Purves, A. Robinson, J. Simpson, S. L. Tuck, E. Weiher, H. J. White, R. M. Ewers, G. M. Mace, J. P. Scharlemann, and A. Purvis. 2015. Global effects of land use on local terrestrial biodiversity. Nature 520:45–.

Newman, M. E., McLaren, K. P., & Wilson, B. S. 2014a. Long-term socio-economic and spatial pattern drivers of land cover change in a Caribbean tropical moist forest, the Cockpit Country, Jamaica. Agriculture, Ecosystems & Environment, 186, 185-200.

Newman, M.E., McLaren, K.P. and Wilson, B.S., 2014b. Assessing deforestation and fragmentation in a tropical moist forest over 68 years; the impact of roads and legal protection in the Cockpit Country, Jamaica. Forest Ecology and Management, 315, pp.138-152.

Newman, S. P., E. H. Meesters, C. S. Dryden, S. M. Williams, C. Sanchez, P. J. Mumby, and N. V. C. Polunin. 2015. Reef flattening effects on total richness and species responses in the Caribbean. Journal of Animal Ecology 84:1678–1689.

Nielsen, J. L. 2014. Recovering the Interior Least Tern: A Fresh Approach to De-listing a Species. Bird Conservation 6–11.

NOAA. 2015. Recovering Threatened and Endangered Species: FY 2013-2014 Report to Congress. National Oceanic and Atmospheric Administration.

NOAA. 2016. Status of Stocks 2015: Annual Report to Congress on the Status of U.S. Fisheries. National Oceanic and Atmospheric Administration.

Nóbrega, G. N., X. L. Otero, F. Macías, and T. O. Ferreira. 2014. Phosphorus geochemistry in a Brazilian semiarid mangrove soil affected by shrimp farm effluents. Environmental Monitoring and Assessment 186:5749-5762.

Nori, J., Urbina-Cardona, J. N., Loyola, R. D., Lescano, J. N., & Leynaud, G. C. (2011). Climate change and American Bullfrog invasion: What could we expect in South America? PLoS ONE, 6(10), 1–8. https://doi.org/10.1371/journal.pone.0025718

Notaro, M., A. Mauss, and J. W. Williams. 2012. Projected vegetation changes for the American Southwest : combined dynamic modeling and bioclimatic-envelope approach. Ecological Applications 22:1365–1388.

Notaro, M., V. Bennington, and B. Lofgren. 2015a. Dynamical downscaling-based projections of Great Lakes' water levels. Journal of Climate In Revisio:9721–9745.

Notaro, M., V. Bennington, and S. Vavrus. 2015b. Dynamically downscaled projections of lake-effect snow in the Great Lakes basin. Journal of Climate 28:1661–1684.

Nunes, Luci. (2011). An Overview of Recent Natural Disasters in South America. Bulletin des séances-Académie royale des sciences d'outre-mer. 57. 409-425.

Nungesser, M., C. Saunders, C. Coronado-Molina, J. Obeysekera, J. Johnson, C. McVoy, and B. Benscoter. 2015. Potential Effects of Climate Change on Florida's Everglades. Environmental Management 55:824–835.

Nurse, L.A., R.F. McLean, J. Agard, L.P. Briguglio, V. Duvat-Magnan, N. Pelesikoti, E. Tompkins, and A. Webb, 2014:Small islands. In: Climate Change 2014: Impacts, Adaptation, and Vulnerability. Part B: Regional Aspects.Contribution of Working Group II to the Fifth Assessment Report of the Intergovernmental Panel on ClimateChange [Barros, V.R., C.B. Field, D.J. Dokken, M.D. Mastrandrea, K.J. Mach, T.E. Bilir, M. Chatterjee, K.L. Ebi,Y.O. Estrada, R.C. Genova, B. Girma, E.S. Kissel, A.N. Levy, S. MacCracken, P.R. Mastrandrea, and L.L. White(eds.)]. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA, pp. 1613-165

Oken, E., Wright, R.O., Kleinman, K.P., Bellinger, D., Amarasiriwardena, C.J., Hu, H., Rich-Edwards, J.W. and Gillman, M.W., 2005. Maternal fish consumption, hair mercury, and infant cognition in a US cohort. Environmental health perspectives, 113(10), p.1376.

Olivares, I., J. C. Svenning, P. M. van Bodegom, and H. Balslev. 2015. Effects of Warming and Drought on the Vegetation and Plant Diversity in the Amazon Basin. Botanical Review 81:42–69.

Oliver, T. H., & Morecroft, M. D. (2014). Interactions between climate change and land use change on biodiversity: attribution problems, risks, and opportunities. Wiley Interdisciplinary Reviews: Climate Change, 5(3), 317-335.

Orensanz JM, Schwindt E, Pastorino G, Bortolus A, Casas G, Darrigran G, Elías R, López Gappa JJ, Obenat S, Pascual M, Penchaszadeh P, Píriz ML, Scarabino F, Spivak E & Vallarino EA. 2002. No longer the pristine confines of the world ocean: a survey of exotic marine species in the southwestern Atlantic. Biological Invasions, 4: 115-143.

Orta-Martínez, M, Napolitano, D. A., MacLennan, G. J., O'Callaghan, C., Ciborowski, S., & Fabregas, X. (2007). Impacts of petroleum activities for the Achuar people of the Peruvian Amazon: summary of existing evidence and research gaps. Environmental Research Letters, 2(4), 045006.

Orta-Martínez, M., and M. Finer. 2010. Oil frontiers and indigenous resistance in the Peruvian Amazon. Ecological Economics 70:207–218.

Paavola, J. (2007). Institutions and environmental governance: A reconceptualization. Ecological Economics, 63(1), 93–103. http://doi.org/10.1016/j.ecolecon.2006.09.026

Pacyna, E.G., Pacyna, J.M., Steenhuisen, F. and Wilson, S., 2006. Global anthropogenic mercury emission inventory for 2000. Atmospheric environment, 40(22), pp.4048-4063.

Padoch, C., E. Brondizio, S. Costa, M. Pinedo-Vasquez, R. R. Sears, and A. Siqueria. 2008. Urban forest and rural cities: Multi-sited households, consumption patterns, and forest resources in Amazonia. Ecology and Society 13.

Painter, T. H., J. S. Deems, J. Belnap, A. F. Hamlet, C. C. Landry, and B. Udall. 2010. Response of Colorado River runoff to dust radiative forcing in snow. Proceedings of the National Academy of Sciences of the United States of America 107:17125–17130.

Palmer, M. A., Moglen, G. E., Bockstael, N. E., Brooks, S., Pizzuto, J. E., Wiegand, C., & VanNess, K. (2002). The ecological consequences of changing land use for running waters, with a case study of urbanizing watersheds in Maryland. Science, 287, 1170-1774.

Pandolfi, J. M., S. R. Connolly, D. J. Marshall, and A. L. Cohen. 2011. Projecting Coral Reef Futures Under Global Warming and Ocean Acidification. Science 333:418–422. <http://science.sciencemag.org/content/333/6041/418\nhttp://science.sciencemag.org/content/sci/333/6041/418.full.pdf\nhttp://www.ncbi.nlm.nih.gov/pubmed/21778392>.

Pardo, L. H., M. E. Fenn, C. L. Goodale, L. H. Geiser, C. T. Driscoll, E. B. Allen, J. S. Baron, R. Bobbink, W. D. Bowman, C. M. Clark, B. Emmett, F. S. Gilliam, T. L. Greaver, S. J. Hall, E. A. Lilleskov, L. Liu, J. A. Lynch, K. J. Nadelhoffer, S. S. Perakis, M. J. Robin-Abbott, J. L. Stoddard, K. C. Weathers, and R. L. Dennis. 2011. Effects of nitrogen deposition and empirical nitrogen critical loads for ecoregions of the United States. Ecological Applications 21:3049–3082.

Paredes, M. (2016). The glocalization of mining conflict: Cases from Peru. Extractive Industries and Society, 3(4), 1046–1057. http://doi.org/10.1016/j.exis.2016.08.007

Paris, C. B., D. Aldana-Aranda, M. P. Perez, and J. Kool. 2008. Connectivity of Queen conch, Strombus gigas, populations from Mexico. Pages 439–443 in. Proceedings of the 11th International Coral Reef Symposium. Ft. Lauderdale.

Parrotta, J. A. (1992). The role of plantation forests in rehabilitating degraded tropical ecosystems. Agriculture, Ecosystems & Environment, 41(2), 115-133.

Parrotta, J. A., Turnbull, J. W., & Jones, N. (1997). Catalyzing native forest regeneration on degraded tropical lands. Forest Ecology and Management, 99(1), 1-7.

Paruelo J. M., Jobbágy, E.G, Oesterheld M. Golluscio R.A. and Aguiar M.R. 2007. The grasslands and steppes of Patagonia and the Rio de la Plata plains. In T. Veblen, K. Young and A. Orme (eds.). Chapter 14. The Physical Geography of South America. The Oxford Regional Environments Series, Oxford University Press. Pp 232-248.

Passos, C. J. S., and D. Mergler. 2008. Human mercury exposure and adverse health effects in the Amazon : a review Exposição humana ao mercúrio e efeitos adversos à saúde na Amazônia : uma revisão. Revisao 24:503–520.

Pastick, N.J., P. Duffy, H. Genet, T.S. Rupp, B.K. Wylie, K.D. Johnson, M.T. Jorgenson, N. Bliss, A.D. McGuire, E.E. Jafarov, and J.F. Knight. 2017. Historical and projected trends in landscape drivers affecting carbon dynamics in Alaska. Ecological Applications. In press.

Pastorino, G., Penchaszadeh, P.E., Schejter, L. and C.S. Bremec. 2000. Rapana venosa (Valenciennes, 1846) (Mollusca: Muricidae): a new gastropod in South Atlantic waters. Journal of Shellfish Research, 19(2): 897-899.

Pastorino, G.; Darrigran, G.; Martin, S.M. & L. Lunaschi, L. 1993. Limnoperna fortunei (Dunker, 1857)(Mytilidae), nuevo bivalvo invasor en aguas del Rio da Prata. Neotropica,39: 34-36.

Patterson, K. L., J. W. Porter, K. B. Ritchie, S. W. Polson, E. Mueller, E. C. Peters, D. L. Santavy, and G. W. Smith. 2002. The etiology of white pox, a lethal disease of the Caribbean elkhorn coral, Acropora palmata. Proceedings of the National Academy of Sciences of the United States of America 99:8725–30. <http://www.pnas.org/content/99/13/8725.short>.

Pauchard A, R García, S Zalba, M Sarasola, R Zenni, S Ziller & MA Nuñez. 2015. Pine Invasions in South America: Reducing Their Ecological Impacts Through Active Management. En: João Canning-Clode (Ed.) Biological Invasions in Changing Ecosystems Vectors, Ecological Impacts, Management and Predictions. p 318 - 342. De Gruyter Open, Berlin.

Pauchard, A., & Barbosa, O. (2013). Regional assessment of Latin America: rapid urban development and social economic inequity threaten biodiversity hotspots. In Urbanization, biodiversity and ecosystem services: Challenges and opportunities (pp. 589-608). Springer Netherlands.

Pauchard, A., Kueffer, C., Dietz, H., Daehler, C. C., Alexander, J., Edwards, P. J., Arévalo, J. R., Cavieres, L. A., Guisan, A., Haider, S., Jakobs, G., McDougall, K., Millar, C. I., Naylor, B. J., Parks, C. G., Rew, L. J., & Seipel, T. (2009). Ain't no mountain high enough: Plant invasions reaching new elevations. Frontiers in Ecology and the Environment, 7(9), 479–486. https://doi.org/10.1890/080072

Pauchard, A., Milbau, A., Albihn, A., Alexander, J., Burgess, T., Daehler, C., Englund, G., Essl, F., Evengard, B., Greenwood, G. B., Haider, S., Lenoir, J., McDougall, K., Muths, E., Nuñez, M. A., Olofsson, J., Pellissier, L., Rabitsch, W., Rew, L. J., Robertson, M., Sanders, N., & Kueffer, C. (2016). Non-native and native organisms moving into high elevation and high latitude ecosystems in an era of climate change: new challenges for ecology and conservation. Biological Invasions, 18(2), 345–353. https://doi.org/10.1007/s10530-015-1025-x

Pauchard, A., Nuñez, M. A., Raffaele, E., Bustamante, R., Ledgard, N. J., Relva, M. A., & Simberloff, D. (2010). Introduced conifer invasion in South America: an update. The Scientific Magazine of the International Biogeography Society, 2(2), 34–36.

Pauly, D. and Palomares, M.L., 2005. Fishing down marine food web: it is far more pervasive than we thought. Bulletin of Marine Science, 76(2), pp.197-212.

Pauly, D., & Palomares, M. (2005). Fishing down marine food web: It is far more persuasive than we thought. Bulletin of Marine Science, 76(2), 197–211. https://doi.org/10.1126/science.279.5352.860

Pawlak, A. R., Mack, R. N., Busch, J. W., & Novak, S. J. (2014). Invasion of Bromus tectorum (L.) into California and the American Southwest: rapid, multi-directional and genetically diverse. Biological Invasions, 17(1), 287–306.

Pelc, R., Max, L., Norden, W., Roberts, S., Silverstein, R., & Wilding, S. (2015). Further action on bycatch could boost United States fisheries performance. Marine Policy, 56, 56–60. https://doi.org/10.1016/J.MARPOL.2015.02.002

Penchaszadeh PE, Boltovskoy D, Borges M, Cataldo D, Damborenea C, et al. (2005) Invasores. Invertebrados exóticos en el Río de la Plata y región marina aledaña. Buenos Aires: Eudeba. pp 377.

Pereira, H. M., Leadley, P., Proença, V., Alkemade, R., Scharlemann, J. P. W., Fernandez-Manjarrés, J. F., Araújo, M. B., Balvanera, P., Biggs, R., Cheung, W. W. L., Chini, L., Cooper, H. D., Gilman, E. L., Guénette, S., Hurtt, G. C., Huntington, H. P., Mace, G. M., Oberdorff, T., Revenga, C., Rodrigues, P., Scholes, R. J., Sumaila, U. R., & Walpole, M. (2010). Scenarios for global biodiversity in the 21st century. Science, 330(6010), 1496–1501. https://doi.org/10.1126/science.1196624

Perelman, S. B., Chaneton, E. J., Batista, W. B., Burkart, S. E., & León, R. J. C. (2007). Habitat stress, species pool size and biotic resistance influence exotic plant richness in the Flooding Pampa grasslands. Journal of Ecology, 95(4), 662–673. https://doi.org/10.1111/j.1365-2745.2007.01255.x

Pérez-Ramírez, M., Castrejón, M., Gutiérrez, N. L., & Defeo, O. (2016). The Marine Stewardship Council certification in Latin America and the Caribbean: A review of experiences, potentials and pitfalls. Fisheries Research, 182, 50–58. http://doi.org/10.1016/j.fishres.2015.11.007

Pérez, C. 2004. Technological Revolutions, Paradigm Shifts and Socio-Institutional Change. Globalization, económic development and inequality an alternative perspective 217–242.

Perner, K., Leipe, T., Dellwig, O., Kuijpers, A., Mikkelsen, N., Andersen, T.J. and Harff, J., 2010. Contamination of arctic Fjord sediments by Pb–Zn mining at Maarmorilik in central West Greenland. Marine pollution bulletin, 60(7), pp.1065-1073.

Peterson, T. C., Heim Jr, R. R., Hirsch, R., Kaiser, D. P., Brooks, H., Diffenbaugh, N. S., ... & Katz, R. W. (2013). Monitoring and understanding changes in heat waves, cold waves, floods, and droughts in the United States: state of knowledge. Bulletin of the American Meteorological Society, 94(6), 821-834.

Phillips, O. L., L. E. O. C. Aragão, S. L. Lewis, J. B. Fisher, J. Lloyd, G. López-gonzález, Y. Malhi, A. Monteagudo, J. Peacock, C. A. Quesada, G. Van Der Heijden, S. Almeida, I. Amaral, L. Arroyo, G. Aymard, R. Tim, O. Bánki, L. Blanc, D. Bonal, P. Brando, J. Chave, Á. Cristina, A. De Oliveira, N. D. Cardozo, C. I. Czimczik, S. Science, N. Series, N. Mar, O. L. Phillips, L. E. C. Aragao, S. L. Lewis, J. B. Fisher, J. Lloyd, G. Lopez-gonzalez, Y. Malhi, A. Monteagudo, J. Peacock, V. Der Heijden, T. R. Baker, O. Ba, A. Cristina, J. Chave, L. Blanc, E. Gloor, N. Higuchi, E. Jimenez, G. Lloyd, P. Meir, K. Chao, T. Erwin, A. Di Fiore, E. H. C, H. Keeling, T. Killeen, W. F. Laurance, A. P. Cruz, and N. C. A. Pitman. 2009. Drought Sensitivity of the Amazon Rainforest. Science 323:1344–1347.

Philpot, S., Hipel, K., & Johnson, P. (2016). Strategic analysis of a water rights conflict in the south western United States. Journal of Environmental Management, 180, 247–256. http://doi.org/10.1016/j.jenvman.2016.05.027

Pichs, Ramon (2008). Cambio climático, globalización y subdesarrollo. Editorial Científico-Técnica, La Habana.

Pichs, Ramón (2012). Recursos naturales, economía mundial y crisis ambiental. Ruth Casa Editorial / Editorial Científico-Técnica, La Habana, 2012

Pimentel, D., Lach, L., Zuniga, R., & Morrison, D. (2000). Environmental and economic costs of nonindigenous species in the United States. Bio Science, 50(1), 53–64.

Pinkerton, E., & Davis, R. (2015). Neoliberalism and the politics of enclosure in North American small-scale fisheries. Marine Policy, 61, 303–312. http://doi.org/10.1016/j.marpol.2015.03.025

Pinto, L. F. G., & McDermott, C. (2013). Equity and forest certification - A case study in Brazil. Forest Policy and Economics, 30, 23–29.

Pirard, R., & Belna, K. (2012). Agriculture and Deforestation: Is REDD+ Rooted in Evidence? Forest Policy and Economics, 21, 62–70. http://doi.org/10.1016/j.forpol.2012.01.012

Pirrone, N., Cinnirella, S., Feng, X., Finkelman, R.B., Friedli, H.R., Leaner, J., Mason, R., Mukherjee, A.B., Stracher, G.B., Streets, D.G. and Telmer, K., 2010. Global mercury emissions to the atmosphere from anthropogenic and natural sources. Atmospheric Chemistry and Physics, 10(13), pp.5951-5964.

Pivello, V. R. (2014). Invasões Biológicas no Cerrado Brasileiro: Efeitos da Introdução de Espécies Exóticas sobre a Biodiversidade. Ecologia. Retrieved from http://www.ecologia.info/cerrado.htm

Pivello, V. R. 2011. The use of fire in the cerrado and Amazonian rainforests of Brazil: Past and present. Fire Ecology 7:24–39.

Poff, N. L., J. D. Allan, M. B. Bain, J. R. Karr, K. L. Prestegaard, B. D. Richter, R. E. Sparks, and J. C. Stromberg. 1997. The Natural Flow Regime. BioScience 47:769–784.

Pokorny, B., Pacheco, P., Cerutti, P.O., van Solinge, T.B., Kissinger, G., & Tacconi, L. (2016). Drivers of Illegal and Destructive Forest Use. In: Kleinschmit, D., Mansourian, S., Wildburger, C., Purret, A., International Union of Forest Research Organizations, G., & Tacconi, L. (2016). Illegal logging and related timber trade : dimensions, drivers, impacts and responses : a global scientific rapid response assessment report. International Union of Forest Research Organizations (IUFRO), World Series no. 35. Vienna, Austria. Retrieved from http://www.cifor.org/library/6312/drivers-of-illegal-and-destructive-forest-use/

Pollock, H. S., Z. A. Cheviron, T. J. Agin, and J. D. Brawn. 2015. Absence of microclimate selectivity in insectivorous birds of the Neotropical forest understory. Biological Conservation 188:116–125. Elsevier Ltd. <http://dx.doi.org/10.1016/j.biocon.2014.11.013>.

Ponce-Reyes, R., Nicholson, E., Baxter, P. W., Fuller, R. A., & Possingham, H. (2013). Extinction risk in cloud forest fragments under climate change and habitat loss. Diversity and Distributions, 19(5-6), 518-529.

Poore, J. A. (2016). Call for conservation: Abandoned pasture. Science, 351(6269), 132-132.

Poorter, L., F. Bongers, T. M. Aide, A. M. Almeyda Zambrano, P. Balvanera, J. M. Becknell, V. Boukili, P. H. S. Brancalion, E. N. Broadbent, R. L. Chazdon, D. Craven, J. S. de Almeida-Cortez, G. A. L. Cabral, B. H. J. de Jong, J. S. Denslow, D. H. Dent, S. J. DeWalt, J. M. Dupuy, S. M. Durán, M. M. Espírito-Santo, M. C. Fandino, R. G. César, J. S. Hall, J. L. Hernandez-Stefanoni, C. C. Jakovac, A. B. Junqueira, D. Kennard, S. G. Letcher, J.-C. Licona, M. Lohbeck, E. Marín-Spiotta, M. Martínez-Ramos, P. Massoca, J. A. Meave, R. Mesquita, F. Mora, R. Muñoz, R. Muscarella, Y. R. F. Nunes, S. Ochoa-Gaona, A. A. de Oliveira, E. Orihuela-Belmonte, M. Peña-Claros, E. A. Pérez-García, D. Piotto, J. S. Powers, J. Rodríguez-Velázquez, I. E. Romero-Pérez, J. Ruíz, J. G. Saldarriaga, A. Sanchez-Azofeifa, N. B. Schwartz, M. K. Steininger, N. G. Swenson, M. Toledo, M. Uriarte, M. van Breugel, H. van der Wal, M. D. M. Veloso, H. F. M. Vester, A. Vicentini, I. C. G. Vieira, T. V. Bentos, G. B. Williamson, and D. M. A. Rozendaal. 2016. Biomass resilience of Neotropical secondary forests. Nature 530:211–214. <http://www.nature.com/doifinder/10.1038/nature16512>.

Pope III, C.A., Ezzati, M. and Dockery, D.W., 2009. Fine-particulate air pollution and life expectancy in the United States. N Engl J Med, 2009(360), pp.376-386.

Porter, E.M., Bowman, W.D., Clark, C.M., Compton, J.E., Pardo, L.H. and Soong, J.L., 2013. Interactive effects of anthropogenic nitrogen enrichment and climate change on terrestrial and aquatic biodiversity. Biogeochemistry, 114(1-3), pp.93-120.

Portillo-Quintero, C. A., & Sánchez-Azofeifa, G. A. (2010). Extent and conservation of tropical dry forests in the Americas. Biological Conservation, 143(1), 144-155.

Possingham, H. P., & Wilson, K. A. (2005). Biodiversity: Turning up the heat on hotspots. Nature, 436(7053), 919-920.

Pozo, K., Urrutia, R., Barra, R., Mariottini, M., Treutler, H.C., Araneda, A. and Focardi, S., 2007. Records of polychlorinated biphenyls (PCBs) in sediments of four remote Chilean Andean Lakes. Chemosphere, 66(10), pp.1911-1921.

Pozo, K., Urrutia, R., Mariottini, M., Rudolph, A., Banguera, J., Pozo, K., Parra, O. and Focardi, S., 2014. Levels of persistent organic pollutants (POPs) in sediments from Lenga estuary, central Chile. Marine pollution bulletin, 79(1), pp.338-341.

Price, D. T., R. I. Alfaro, K. J. Brown, M. D. Flannigan, R. A. Fleming, E. H. Hogg, M. P. Girardin, T. Lakusta, M. Johnston, D. W. Mckenney, J. H. Pedlar, T. Stratton, R. N. Sturrock, I. D. Thompson, J. A. Trofymow, and L. A. Venier. 2013. Anticipating the consequences of climate change for Canada's boreal forest ecosystems. Environmental Reviews 365:322–365. <http://www.nrcresearchpress.com/doi/full/10.1139/er-2013-0042>.

Prieto-Torres, D. A., A. G. Navarro-Sig??enza, D. Santiago-Alarcon, and O. R. Rojas-Soto. 2015. Response of the endangered tropical dry forests to climate change and the role of Mexican Protected Areas for their conservation. Global Change Biology 364–379.

Purcell, S. W., A. Mercier, C. Conand, J. F. Hamel, M. V. Toral-Granda, A. Lovatelli, and S. Uthicke. 2013. Sea cucumber fisheries: Global analysis of stocks, management measures and drivers of overfishing. Fish and Fisheries 14:34–59.

Pyne, S., Andrews, P.L., Laven, R. D. 1996. Introduction to Wildland Fire. John Wiley & Sons, Inc, .

Pyšek, P., Jarošík, V., Pergl, J., Randall, R., Chytrý, M., Kühn, I., Tichý, L., Danihelka, J., Chrtek Jun, J., & Sádlo, J. (2009). The global invasion success of Central European plants is related to distribution characteristics in their native range and species traits. Diversity and Distributions, 15(5), 891–903. https://doi.org/10.1111/j.1472-4642.2009.00602.x

Rabalais, N. N., Turner, R. E., Justić, D., Dortch, Q., Wiseman, W. J., & Gupta, B. K. S. (1996). Nutrient changes in the Mississippi River and system responses on the adjacent continental shelf. Estuaries, 19(2), 386-407.

Rabalais, N.N., Cai, W.J., Carstensen, J., Conley, D.J., Fry, B., Hu, X., Quinones-Rivera, Z., Rosenberg, R., Slomp, C.P., Turner, R.E. and Voss, M., 2014. Eutrophication-driven deoxygenation in the coastal ocean. Oceanography, 27(1), pp.172-183.

Rajotte, Tasmin. 2008. The Negotiations Web: Complex Connections. In Geoff

Ramirez-Villegas, J., F. Cuesta, C. Devenish, M. Peralvo, A. Jarvis, and C. A. Arnillas. 2014. Using species distributions models for designing conservation strategies of Tropical Andean biodiversity under climate change. Journal for Nature Conservation 22:391–404. Elsevier GmbH. <http://dx.doi.org/10.1016/j.jnc.2014.03.007>.

Ramírez, V. M., Ayala, R., & González, H. D. (2016). Temporal variation in native bee diversity in the tropical sub-deciduous forest of the Yucatan Peninsula, Mexico. Tropical Conservation Science, 9(2), 718-734.

Ramos-Scharran, C. E., D. Torres-Pulliza, and E. A. Hernandez-Delgado. 2015. Watershed- and island wide-scale land cover changes in Puerto Rico (1930s-2004) and their potential effects on coral reef ecosystems. Science of the Total Environment 506-507:241–251.

Rankin, D. Freshwater Ecosystems and Human Populations: Great Lakes Case Study. Yale School of Forestry & Environmental Studies Bulletin Series 107, 2002.

Rathmann, R., Szklo, A., & Schaeffer, R. (2010). Land use competition for production of food and liquid biofuels: An analysis of the arguments in the current debate. Renewable Energy, 35(1), 14-22.

Reátegui-Zirena, E.G., Stewart, P.M., Whatley, A., Chu-Koo, F., Sotero-Solis, V.E., Merino-Zegarra, C. and Vela-Paima, E., 2014. Polycyclic aromatic hydrocarbon concentrations, mutagenicity, and Microtox® acute toxicity testing of Peruvian crude oil and oil-contaminated water and sediment. Environmental monitoring and assessment, 186(4), pp.2171-2184.

Rebellato, L., W. I. Woods, and E. G. Neves. 2009. Pre-Columbian settlement dynamics in the Central Amazon. In W. I.Woods, W. G.Teixeira, J.Lehmann, C.Steiner, A. M. G. A.WinklerPrins, and L.Rebellato (Eds.). Amazonian dark earths: Wim Sombroek's vision, pp. 15–32. Springer, Berlin, Germany.

Redmore, L., Stronza, A., Songhurst, A., & McCulloch, G. (2017). Which Way Forward? Past and New Perspectives on Community-Based Conservation in the Anthropocene BT - Reference Module in Earth Systems and Environmental Sciences. Elsevier. http://doi.org/https://doi.org/10.1016/B978-0-12-409548-9.09838-9

Reenberg, A., Langanke, T., Kristensen, S. B. P., & Colding, T. S. (2010). Globalisation of agricultural landscapes a land systems approach. In J. Primdahl & S. Swaffield (Eds.), Globalisation and agricultural landscapes. Change patterns and policy trends in developed countries (pp. 31–56). Cambridge: Cambridge University Press.

Reis, A., Bechara, F. C., Espíndola, M. B. De, Vieira, N. K., & Souza, L. L. De. (2003). Restauração de áreas degradadas : a nucleação como base para incrementar os processos sucessionais. Natureza E Conservação, 1(1), 28–36.

Reis, C. R. G., G. B. Nardoto, and R. S. Oliveira. 2017. Global overview on nitrogen dynamics in mangroves and consequences of increasing nitrogen availability for these systems. Plant and

Restrepo, J. C., Gruber, C. G., & Machuca, C. M. (2009, June). Energy profile aware routing. In Communications Workshops, 2009. ICC Workshops 2009. IEEE International Conference on (pp. 1-5). IEEE.

Revenga, C., Campbell, I., Abell, R., De Villiers, P., & Bryer, M. (2005). Prospects for monitoring freshwater ecosystems towards the 2010 targets. Philosophical Transactions of the Royal Society of London B: Biological Sciences, 360(1454), 397-413.

Reyer, C. P. O., Adams, S., Albrecht, T., Baarsch, F., Boit, A., Trujillo, N. C., Cartsburg, M., Coumou, D., Eden, A., Fernandes, E., Langerwisch, F., Marcus, R., Mengel, M., Mira-Salama, D., Perette, M., Pereznieto, P., Ramirez-Villegas, J., Reinhardt, J., Robinson, A., Rocha, M., Sakschewski, B., Schaeffer, M., Schleussner, C.-F., Serdeczny, O., & Thonicke, K. (2015). Climate change impacts in Latin America and the Caribbean and their implications for development. Regional Environmental Change, 1–21. https://doi.org/10.1007/s10113-015-0854-6

Ribeiro, K.H., Favaretto, N., Dieckow, J., Souza, L.C.D.P., Minella, J.P.G., Almeida, L.D. and Ramos, M.R., 2014. Quality of surface water related to land use: a case study in a catchment with small farms and intensive vegetable crop production in southern Brazil. Revista Brasileira de Ciência do Solo, 38(2), pp.656-668.

Ricaurte, L. F., Wantzen, K. M., Agudelo, E., Betancourt, B., & Jokela, J. (2014). Participatory rural appraisal of ecosystem services of wetlands in the Amazonian Piedmont of Colombia: Elements for a sustainable management concept. Wetlands Ecology and Management, 22(4), 343–361. https://doi.org/10.1007/s11273-013-9333-3

Richards, M., Wells, A., Del Gatto, F., Contreras-Hermosilla, A. and Pommier, D., 2003. Impacts of illegality and barriers to legality: a diagnostic analysis of illegal logging in Honduras and Nicaragua. International Forestry Review, 5(3), pp.282-292.

Richards, P. D., Myers, R. J., Swinton, S. M., & Walker, R. T. (2012). Exchange rates, soybean supply response, and deforestation in South America. Global environmental change, 22(2), 454-462.

Rigét, F., Bignert, A., Braune, B., Stow, J. and Wilson, S., 2010. Temporal trends of legacy POPs in Arctic biota, an update. Science of the Total Environment, 408(15), pp.2874-2884.

Rignot, E., I. Velicogna, M. R. van den Broeke, A. Monaghan, and J. Lenaerts (2011), Acceleration of the contribution of the Greenland and Antarctic ice sheets to sea level rise, Geophys. Res. Lett., 38, L05503, doi:10.1029/2011GL046583

Ríos, A.F., Resplandy, L., García-Ibáñez, M.I., Fajar, N.M., Velo, A., Padin, X.A., Wanninkhof, R., Steinfeldt, R., Rosón, G. and Pérez, F.F., 2015. Decadal acidification in the water masses of the Atlantic Ocean. Proceedings of the National Academy of Sciences, 112(32), pp.9950-9955.

Ríos, M., Zaldúa, N., & Cupeiro, S. (2010). Evaluación participativa de plaguicidas en el sitio RAMSAR, Parque Nacional Esteros de Farrapos e Islas del Río Uruguay. Vida Silvestre Uruguay, Montevideo.

Ripple, W. J., T. M. Newsome, C. Wolf, R. Dirzo, K. T. Everatt, M. Galetti, M. W. Hayward, G. I. H. Kerley, T. Levi, P. A. Lindsey, D. W. Macdonald, Y. Malhi, L. E. Painter, C. Sandom, J. Terborgh, and B. Van Valkenburgh. 2015. Collapse of the world's largest herbivores. Science Advances 1:1–12.

Rizzo, D. M., & Garbelotto, M. (2003). Sudden oak death: endangering California and Oregon forest ecosystems. Frontiers in Ecology and the Environment, 1(5), 197–204. https://doi.org/10.1890/1540-9295(2003)001[0197:SODECA]2.0.CO;2

Robeson, S. M. 2002. Increasing growing-season length in Illinois during the 20th century. Climatic Change 52:219–238.

Rockström, J., Steffen, W., Noone, K., Persson, Å., Chapin, F.S., Lambin, E.F., Lenton, T.M., Scheffer, M., Folke, C., Schellnhuber, H.J. and Nykvist, B., 2009. A safe operating space for humanity. nature, 461(7263), pp.472-475.

Rodil, I. F., Lucena-Moya, P., Olabarria, C., & Arenas, F. (2015). Alteration of macroalgal subsidies by climate-associated stressors affects behavior of wrack-reliant beach consumers. Ecosystems, 18(3), 428-440.

Rodrigues Capítulo, A., Gómez, N., Giorgi, A. et al., 2010. Global changes in pampean lowland streams (Argentina): Implications for biodiversity and functioning. Hydrobiologia 657:53–70.

Rodrigues, R. R., Gandolfi, S., Nave, A. G., Aronson, J., Barreto, T. E., Vidal, C. Y., & Brancalion, P. H. (2011). Large-scale ecological restoration of high-diversity tropical forests in SE Brazil. Forest Ecology and Management, 261(10), 1605-1613

Rodrigues, R. R., R. A. F. Lima, S. Gandolfi, and A. G. Nave. 2009. On the restoration of high diversity forests: 30 years of experience in the Brazilian Atlantic Forest. Biological Conservation 142:1242–1251. Elsevier Ltd. <http://dx.doi.org/10.1016/j.biocon.2008.12.008>.

Rodríguez-Rodríguez, J. A., P. C. Sierra-Correa, M. C. Gómez-Cubillos, and L. V. L. Villanueva. 2016. Mangrove ecosystems (Colombia). Pages 1-10 in. C. M. Finlayson, M. Everard, K. Irvine, R. J. McInnes, B. Middleton, A. van Dam, and N. C. Davidson, editors. The Wetland Book.

Rojas-Sandoval, J., & Acevedo-Rodríguez, P. (2014). Naturalization and invasion of alien plants in Puerto Rico and the Virgin Islands. Biological Invasions, 17(1), 149–163. https://doi.org/10.1007/s10530-014-0712-3

Romero-Lankao, P., J.B. Smith, D.J. Davidson, N.S. Diffenbaugh, P.L. Kinney, P. Kirshen, P. Kovacs, and L. Villers Ruiz, 2014: North America. In: Climate Change 2014: Impacts, Adaptation, and Vulnerability. Part B: Regional Aspects. Contribution of Working Group II to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change [Barros, V.R., C.B. Field, D.J. Dokken, M.D. Mastrandrea, K.J. Mach, T.E. Bilir, M. Chatterjee, K.L. Ebi,Y.O. Estrada, R.C. Genova, B. Girma, E.S. Kissel, A.N. Levy, S. MacCracken, P.R. Mastrandrea, and L.L. White (eds.)]. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA, pp. 143

Ropelewski, C. F., and M. S. Halpert. 1987. Global and Regional Scale Precipitation Patterns Associated with the El Niño/Southern Oscillation. Monthly Weather Review. Volume 115.

Rossi, R. D., Martins, C. R., Viana, P. L., Rodrigues, E. L., & Figueira, J. E. C. (2014). Impact of invasion by molasses grass (Melinis minutifloraP. Beauv.) on native species and on fires in areas of campo-cerrado in Brazil. Acta Botanica Brasilica, 28(4), 631–637. https://doi.org/10.1590/0102-33062014abb3390

Rozzi, R., Armesto, J. J., Goffinet, B., Buck, W., Massardo, F., Silander, J., Arroyo, M. T. K., Russell, S., Anderson, C. B., Cavieres, L. A., & Callicott, J. B. (2008). Changing lenses to assess biodiversity: Patterns of species richness in sub-Antarctic plants and implications for global conservation. Frontiers in Ecology and the Environment. https://doi.org/10.1890/070020

Rudel, T. K., Defries, R., Asner, G. P., & Laurance, W. F. (2009). Changing drivers of deforestation and new opportunities for conservation. Conservation Biology, 23(6), 1396-1405.

Ruiz, G. M., Rawlings, T. K., Dobbs, F. C., Drake, L. a, Mullady, T., Huq, a, & Colwell, R. R. (2000). Global spread of microorganisms by ships. Nature, 408(6808), 49–50. https://doi.org/10.1038/35040695

Ruiz-Toledo, J., Castro, R., Rivero-Pérez, N., Bello-Mendoza, R. and Sánchez, D., 2014. Occurrence of glyphosate in water bodies derived from intensive agriculture in a tropical region of southern Mexico. Bulletin of environmental contamination and toxicology, 93(3), pp.289-293.

Runde & Magpile, 2014. Science, technology, and innovation as drivers of development: http://csis.org/publication/science-technology-and-innovation-drivers-development

Rupp, T. S., P. Duffy, M. Leonawicz, M. Lindgren, A. Breen, T. Kurkowski, A. Floyd, A. Bennett, and L. Krutikov. 2016. Chapter 2 . Climate Simulations, Land Cover, and Wildfire. Pages 17–52 in Z. Zhu and A. D. McGuire, editors. Baseline and projected future carbon storage and greenhouse-gas fluxes in ecosystems of Alaska.

Rushton, J., R. Viscarra, C. Viscarra, F. Basset, R. Baptista, and D. Brown. 2005. How Important is Bushmeat Consumption in South America: Now and in the Future? Odi Wildlife Policy Briefing 11:1–4.

Ruviaro, C. F., da Costa, J. S., Florindo, T. J., Rodrigues, W., de Medeiros, G. I. B., & Vasconcelos, P. S. (2016). Economic and environmental feasibility of beef production in different feed management systems in the Pampa biome, southern Brazil. Ecological Indicators, 60, 930-939.

Ruyle L.E. (2017). The Impacts of Conflict on Biodiversity in the Anthropocene. Reference Module in Earth Systems and Environmental Sciences. Published on line. doi.org/10.1016/B978-0-12-409548-9.09849-3

Ryan, K. C., Knapp, E. E., & Varner, J. M. (2013). Prescribed fire in North American forests and woodlands: history, current practice, and challenges. Frontiers in Ecology and the Environment, 11(s1).

Saatchi, S., S. Asefi-Najafabady, Y. Malhi, L. E. O. C. Aragão, L. O. Anderson, R. B. Myneni, and R. Nemani. 2013. Persistent effects of a severe drought on Amazonian forest canopy. Proceedings of the National Academy of Sciences of the United Statesof America 110:565–70.

Sala, O. E., & Paruelo, J. M. (1997). Ecosystem services in grasslands. Nature's services: Societal dependence on natural ecosystems, 237-251.

Sala, O. E., F. S. Chapin III, J. J. Armesto, E. Berlow, J. Bloomfield, R. Dirzo, E. Huber-Sanwald, L. F. Huenneke, R. B. Jackson, A. Kinzig, R. Leemans, D. M. Lodge, H. A. Mooney, M. Oesterheld, N. L. Poff, M. T. Skykes, B. H. Walker, M. Walker, and D. H. Wall. (2000). Global Biodiversity Scenarios for the Year 2100. Science 287:1770 – 1774.

Sales, I. dos S., Ruiz-Miranda, C. R., & Santos, C. de P. (2010). Helminths found in marmosets (Callithrix penicillata and Callithrix jacchus) introduced to the region of occurrence of golden lion tamarins (Leontopithecus rosalia) in Brazil. Veterinary Parasitology, 171(1–2), 123–129. https://doi.org/10.1016/j.vetpar.2010.02.044

Salles, J., & Correa, R. (2004). Água de Lastro e Bioinvasão. Interciência.

San Sebastián, M., and A.-K. Hurtig. (2004). Oil exploitation in the Amazon basin of Ecuador: a public health emergency. Revista panamericana de salud publica = Pan American journal of public health 15:205–211.

San Sebastian, M., Armstrong, B., & Stephens, C. (2002). Outcomes of pregnancy among women living in the proximity of oil fields in the Amazon basin of Ecuador. International Journal of Occupational and Environmental Health, 8(4), 312–319. https://doi.org/10.1179/107735202800338650

Sánchez-Azofeifa, A., Powers, S.J., Fernandez, G.W., Quesada, Q. Tropical Dry Forests in the Americas: Ecology, Conservation, and Management CRC Press, Taylor & Francis Group, Boca Raton, USA (2014)

Sánchez-Azofeifa, G. A., Castro-Esau, K. L., Kurz, W. A., & Joyce, A. (2009). Monitoring carbon stocks in the tropics and the remote sensing operational limitations: from local to regional projects. Ecological Applications, 19(2), 480-494.

Sánchez, A., Ortiz-Hernández, M. C., Talavera-Sáenz, A., & Aguíñiga-García, S. (2013). Stable nitrogen isotopes in the turtle grass Thalassia testudinum from the Mexican Caribbean: implications of anthropogenic development. Estuarine, Coastal and Shelf Science, 135, 86-93.

Sandoval-Herrera, N.I., Vargas-Soto, J.S., Espinoza, M., Clarke, T.M., Fisk, A.T. and Wehrtmann, I.S., 2016. Mercury levels in muscle tissue of four common elasmobranch species from the Pacific coast of Costa Rica, Central America. Regional Studies in Marine Science, 3, pp.254-261.

Santo, A.R., M.G. Sorice, C.J. Donlan, C.T. Franck & C.B. Anderson (2015). A human-centered approach to designing invasive species eradication programs on human-inhabited islands. Global Environmental Change 35: 289-298.

Santos-Santos, E., Yarto-Ramírez, M., Gavilán-García, I., Castro-Díaz, J., Gavilán-García, A., Rosiles, R., Suárez, S. and López-Villegas, T., 2006. Analysis of arsenic, lead and mercury in farming areas with mining contaminated soils at Zacatecas, Mexico. Journal of the Mexican Chemical Society, 50(2), pp.57-63.

Santos, C. A. D. (2014). Recent changes in temperature and precipitation extremes in an ecological reserve in federal district, Brazil. Revista Brasileira de Meteorologia, 29(1), 13–20. https://doi.org/10.1590/S0102-77862014000100002

Santos, H. F., F. L. Carmo, J. E. S. Paes, A. S. Rosado, and R. S. Peixoto. 2011. Bioremediation of Mangroves Impacted by Petroleum. Water, Air, & Soil Pollution 216:329–350.

Sarukhán, J., Urquiza-Haas, T., Koleff, P., Carabias, J., Dirzo, R., Ezcurra, E., Cerdeira-Estrada, S., & Soberón, J. (2015). Strategic Actions to Value, Conserve, and Restore the Natural Capital of Megadiversity Countries: The Case of Mexico. BioScience, 65(2), 164–173. https://doi.org/10.1093/biosci/biu195

Satyamurty, P., A. A. De Castro, J. Tota, L. E. Da Silva Gularte, and A. O. Manzi. 2010. Rainfall trends in the Brazilian Amazon Basin in the past eight decades. Theoretical and Applied Climatology 99:139–148.

Savini, D. & Occhipinti-Ambrogi, A. 2006. Consumption rates and prey preference of the invasive gas-tropod Rapana venosa in the Northern Adriatic Sea. Helgoland Marine Research, 60: 153-159.

Sayer, E.J., Wright, S.J., Tanner, E.V., Yavitt, J.B., Harms, K.E., Powers, J.S., Kaspari, M., Garcia, M.N. and Turner, B.L., 2012. Variable responses of lowland tropical forest nutrient status to fertilization and litter manipulation. Ecosystems, 15(3), pp.387-400.

Scanlon, B. R., I. Jolly, M. Sophocleous, and L. Zhang. 2007. Global impacts of conversions from natural to agricultural ecosystems on water resources: Quantity versus quality. Water Resources Research 43.

Scarabino F, Menafra R and Etchegaray P.1999. Presencia de Rapana venosa (Valenciennes, 1846) (Gastropoda: Muricidae) en el Río de la Plata. Actas V Jornadas de Zoología del Uruguay, Bo-letín de la Sociedad Zoológica del Uruguay (Segunda época), 11: 40.

Scatena, F. N., and M. C. Larsen. 1991. Physical aspects of Hurricane Hugo in Puerto Rico. Biotropica 23:317–323.

Schaeffer-Novelli, Y., E. J. Soriano-Serra, C. C. Vale, E. Bernini, A. S. Rovai, M. A. A. Pinheiro, A. J. Schmidt, R. Almeida, Jr. Coelho C., R. P. Menghini, D. I. Martinez, G. M. O. Abuchahla, M. Cunha-Lignon, S. Charlier-Sarubo, J. Shirazawa-Freitas, G. Cintrón-Molero. 2016. Climate changes in mangrove forests and salt marshes. Brazilian Journal of Oceanography 64: 37-52.

Scheffers, B. R., L. De Meester, T. C. L. Bridge, A. A. Hoffmann, J. M. Pandolfi, R. T. Corlett, S. H. M. Butchart, P. Pearce-Kelly, K. M. Kovacs, D. Dudgeon, M. Pacifici, C. Rondinini, W. B. Foden, T. G. Martin, C. Mora, D. Bickford, and J. E. M. Watson. 2016. The broad footprint of climate change from genes to biomes to people. Science 354:aaf7671.

Schindler, D.W., Dillon, P.J. and Schreier, H., 2006. A review of anthropogenic sources of nitrogen and their effects on Canadian aquatic ecosystems. Biogeochemistry, 79(1), pp.25-44.

Schleupner, C., & Link, P. M. (2008). Potential impacts on important bird habitats in Eiderstedt (Schleswig-Holstein) caused by agricultural land use changes. Applied Geography, 28(4), 237-247.

Schmidtko, S., Stramma, L. and Visbeck, M., 2017. Decline in global oceanic oxygen content during the past five decades. Nature, 542(7641), pp.335-339.

Schneider, L. (2006). Invasive Species and Land-Use: The Effect of Land Management Practices on Bracken Fern Invasion in the Region of Calakmul, Mexico. Journal of Latin American Geography, 5(2), 92–107.

Schneider, L. C., & Fernando, D. N. (2010). An untidy cover: Invasion of bracken fern in the shifting cultivation systems of southern yucat??n, Mexico. Biotropica, 42(1), 41–48. https://doi.org/10.1111/j.1744-7429.2009.00569.x

Schofield, P. J. (2010). Update on geographic spread of invasive lionfishes (Pterois volitans [Linnaeus, 1758] and P. miles [Bennett, 1828]) in the Western North Atlantic Ocean, Caribbean Sea and Gulf of Mexico. Aquatic Invasions, 5(SUPPL. 1), 117–122. https://doi.org/10.3391/ai.2010.5.S1.024

Schofield, P.J. 2009. Geographic extent and chronology of the invasion of non-native lionfish (Pterois volitans [Linnaeus 1758] and P. miles [Bennett 1828]) in the Western North Atlantic and Caribbean Sea. Aquatic Invasions 4: 473c InvaSchofield, P. J. 2010. Update on geographic spread of invasive lionfishes (Pterois volitans [Linnaeus, 1758] and P. miles [Bennett, 1828]) in the Western North Atlantic Ocean, Caribbean Sea and Gulf of Mexico. Aquatic Invasions 5: S117-S122.

Schöngart, J., F. Wittmann, and M. Worbes. 2010. Biomass and net primary production of central Amazonian floodplain forests. Pages 347–388 in. Amazonian Floodplain Forests: ecophysiology, biodiversity and sustainable management. Volume 210. <http://dx.doi.org/10.1007/978-90-481-8725-6_18>.

Schwarzenbach, R. P., Escher, B. I., Fenner, K., Hofstetter, T. B., Johnson, C. A., Von Gunten, U., & Wehrli, B. (2006). The challenge of micropollutants in aquatic systems. Science, 313(5790), 1072-1077.

Seabloom W. Eric, Williams W. John, Slayback Daniel, Stoms M. David, V. H. J. & D. P. A. (2006). Human Impacts, Plant Invasion, and Imperiled Plant Species in California. Ecological Applications, 16(4), 1338–1350.

Seebens, H., Blackburn, T. M., Dyer, E. E., Genovesi, P., Hulme, P. E., Jeschke, J. M., Pagad, S., Pyšek, P., Winter, M., Arianoutsou, M., Bacher, S., Blasius, B., Brundu, G., Capinha, C., Celesti-Grapow, L., Dawson, W., Dullinger, S., Fuentes, N., Jäger, H., Kartesz, J., Kenis, M., Kreft, H., Kühn, I., Lenzner, B., Liebhold, A., Mosena, A., Moser, D., Nishino, M., Pearman, D., Pergl, J., Rabitsch, W., Rojas-Sandoval, J., Roques, A., Rorke, S., Rossinelli, S., Roy, H. E., Scalera, R., Schindler, S., Štajerová, K., Tokarska-Guzik, B., van Kleunen, M., Walker, K., Weigelt, P., Yamanaka, T., & Essl, F. (2017). No saturation in the accumulation of alien species worldwide. Nature Communications, 8, 14435. https://doi.org/10.1038/ncomms14435

Seebens, H., Gastner, M. T., & Blasius, B. (2013). The risk of marine bioinvasion caused by global shipping. Ecology letters, 16(6), 782-790.

Seijas, A. E., R. Antelo, J. B. Thorbjarnarson, and M. C. Ardila Robayo. 2010. Orinoco Crocodile Crocodylus intermedius. Pages 59–65 in S. C. Manolis and C. Stevenson, editors. Crocodiles. Status Survey and Conservation Action Plan. Third. Crocodile Specialist Group: Darwin.

Seipel, T., Kueffer, C., Rew, L. J., Daehler, C. C., Pauchard, A., Naylor, B. J., Alexander, J. M., Edwards, P. J., Parks, C. G., Arevalo, J. R., Cavieres, L. A., Dietz, H., Jakobs, G., Mcdougall, K., Otto, R., & Walsh, N. (2012). Processes at multiple scales affect richness and similarity of non-native plant species in mountains around the world. Global Ecology and Biogeography, 21(2), 236–246. https://doi.org/10.1111/j.1466-8238.2011.00664.x

Sell, S. K. (2009). Private Power, Public Law. The Globalization of Intellectual Property Rights. Cambridge University Press. Online publication date: September 2009: https://doi.org/10.1017/CBO9780511491665

Serenari, C., Peterson, M. N., Wallace, T., Stowhas, P. (2017). Indigenous Perspectives on Private Protected Areas in Chile. Natural Areas Journal, 37(1), 98–107.

SERNA 2009 https://www.scribd.com/document/98703779/Inventario-de-Humedales-de-Honduras#

Seto, K. C., Fragkias, M., Güneralp, B., & Reilly, M. K. (2011). A meta-analysis of global urban land expansion. PloS one, 6(8), e23777.

Seto, K. C., Güneralp, B., & Hutyra, L. R. (2012). Global forecasts of urban expansion to 2030 and direct impacts on biodiversity and carbon pools. Proceedings of the National Academy of Sciences, 109(40), 16083-16088.

Shelton, Peter A., and M. Joanne Morgan. "Impact of maximum sustainable yield-based fisheries management frameworks on rebuilding North Atlantic cod stocks." Journal of Northwest Atlantic Fishery Science 46 (2014).

Shen, L., Wania, F., Lei, Y.D., Teixeira, C., Muir, D.C. and Xiao, H., 2006. Polychlorinated biphenyls and polybrominated diphenyl ethers in the North American atmosphere. Environmental pollution, 144(2), pp.434-444.

Sherman, K., and Hamukuaya, H. 2016. Sustainable development of the world's Large Marine Ecosystems. Environmental Development. Volume 17, Supplement 1, Pages 1-6.

Sherman, K., Sissenwine, M., Christensen, V., Duda, A., Hempel, G., Ibe, C., Levin, S., et al., 2005. A global movement toward an ecosystem approach to marine resources management. Marine Ecology Progress Series, 300: 275–279.

Shirey, P. D., B. N. Kunycky, D. T. Chaloner, M. A. Brueseke, and G. A. Lamberti. 2013. Commercial trade of federally listed threatened and endangered plants in the United States. Conservation Letters 6:300–316.

Shunthirasingham, C., Gouin, T., Lei, Y.D., Ruepert, C., Castillo, L.E. and Wania, F., 2011. Current-use pesticide transport to Costa Rica's high-altitude tropical cloud forest. Environmental Toxicology and Chemistry, 30(12), pp.2709-2717.

Sieg, C. H. 1987. Small mammals: pests or vital components of the ecosystem. Great Plains Wildlife Damage Control Workshop 97:88–92.

Siffredi, G., López, D., Ayesa, J., Bianchi, E., Velasco, V., & Becker, G. (2011). Reducción de la accesibilidad al forraje por caída de cenizas volcánicas. Revista Presencia, (57).

Silva, E. (2016). Patagonia, without Dams! Lessons of a David vs. Goliath campaign. Extractive Industries and Society, 3(4), 947–957. http://doi.org/10.1016/j.exis.2016.10.004

Silva, J. S. O., da Cunha Bustamante, M. M., Markewitz, D., Krusche, A. V., & Ferreira, L. G. (2011). Effects of land cover on chemical characteristics of streams in the Cerrado region of Brazil. Biogeochemistry, 105(1-3), 75-88.

Silver, W.L., Lugo, A.E. and Keller, M., 1999. Soil oxygen availability and biogeochemistry along rainfall and topographic gradients in upland wet tropical forest soils. Biogeochemistry, 44(3), pp.301-328.

Silvério, D. V., Brando, P. M., Balch, J. K., Putz, F. E., Nepstad, D. C., Oliveira-Santos, C., & Bustamante, M. M. C. (2013). Testing the Amazon savannization hypothesis: fire effects on invasion of a neotropical forest by native cerrado and exotic pasture grasses. Philosophical Transactions of Royal Society B, 8. https://doi.org/10.1098/rstb.2012.0427

Simberloff, D., Martin, J. L., Genovesi, P., Maris, V., Wardle, D. A., Aronson, J., Courchamp, F., Galil, B., García-Berthou, E., Pascal, M., Pyšek, P., Sousa, R., Tabacchi, E., & Vilà, M. (2013). Impacts of biological invasions: What's what and the way forward. Trends in Ecology and Evolution, 28(1), 58–66. https://doi.org/10.1016/j.tree.2012.07.013

Skalak, K. J., Benthem, A. J., Schenk, E. R., Hupp, C. R., Galloway, J. M., Nustad, R. A., & Wiche, G. J. (2013). Large dams and alluvial rivers in the Anthropocene: The impacts of the Garrison and Oahe Dams on the Upper Missouri River. Anthropocene, 2, 51-64.

Smith, R., Muir, R. D., Walpole, M., Balmford, A., & Leader-Williams, N. (2003). Governance and the loss of biodiversity. Nature, 426, 67–70. http://doi.org/10.1038/nature02095.1.

Soares-Filho, B. S., Nepstad, D., Curran, L., Voll, E., Cerqueira, G. C., Garcia, R. A., Ramos, C. A., Mcdonald, A., Lefebvre, P., & Schlesinger, P. (2006). Modeling Conservation in the Amazon Basin. Nature, 440(3), 520–523. Retrieved from https://search.proquest.com/openview/4c0cd9dd431a303d213f9da5edf8a0b1/1?pq-origsite=gscholar&cbl=40569

Sobota, D. J., J. E. Compton, and J. A. Harrison. 2013. Reactive nitrogen inputs to US lands and waterways: How certain are we about sources and fluxes? Frontiers in Ecology and the Environment 11:82–90.

Somers, B., G. P. Asner, R. E. Martin, C. B. Anderson, D. E. Knapp, S. J. Wright, and R. Van De Kerchove. 2015. Mesoscale assessment of changes in tropical tree species richness across a bioclimatic gradient in

Panama using airborne imaging spectroscopy. Remote Sensing of Environment 167:111–120. Elsevier Inc. <http://dx.doi.org/10.1016/j.rse.2015.04.016>.

Son, J. Y., Lee, H. J., Koutrakis, P., & Bell, M. L. (2017). Pregnancy and Lifetime Exposure to Fine Particulate Matter and Infant Mortality in Massachusetts, 2001–2007. American Journal of Epidemiology, 1-9.

Sorensson, A. A., Menéndez, C. G., Ruscica, R., Alexander, P., Samuelsson, P., & Willén, U. (2010). Projected precipitation changes in South America: a dynamical downscaling within CLARIS. Meteorologische Zeitschrift, 19(4), 347-355.

Speziale, K. L., Lambertucci, S. A., Carrete, M., & Tella, J. L. (2012). Dealing with non-native species: What makes the difference in South America? Biological Invasions, 14(8), 1609–1621. https://doi.org/10.1007/s10530-011-0162-0

Sprague, L. A., R. M. Hirsch, and B. T. Aulenbach. 2011. Nitrate in the mississippi river and its tributaries, 1980 to 2008: Are we making progress? Environmental Science and Technology. Volume 45.

Steiner, N.S., Christian, J.R., Six, K.D., Yamamoto, A. and Yamamoto-Kawai, M., 2014. Future ocean acidification in the Canada Basin and surrounding Arctic Ocean from CMIP5 earth system models. Journal of Geophysical Research: Oceans, 119(1), pp.332-347.

Sténs, A., Bjärstig, T., Nordström, E. M., Sandström, C., Fries, C., & Johansson, J. (2016). In the eye of the stakeholder: The challenges of governing social forest values. Ambio, 45, 87–99. http://doi.org/10.1007/s13280-015-0745-6

Stephenson P., Bowles, N., Regan, E., Stanwell, D., Diagana, M., Hoft, R., Abarchi, H., Abrehamse, T., Akello, C., Allison, H., Banki, O., Batieno, B., Dieme, S., Domingos, A., Galt, R., Githaiga, C., Bine, A., Hafashimana, D., & Thiombiano, A. (2017). Unblocking the flow of biodiversity data for decision-making in Africa. Biological Conservation, 213, 335–340. https://doi.org/10.1016/J.BIOCON.2016.09.003

Stocker, T.F., D. Qin, G.-K. Plattner, M. Tignor, S. K. Allen, J. Boschung, A. Nauels, Y. Xia, V. Bex and P.M. Midgley (eds.) (2013 Summary for Policymakers. In: Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Stohlgren, T. J., Jarnevich, C., Chong, G. W., & Evangelista, P. H. (2006). Scale and plant invasions: A theory of biotic acceptance. Preslia, 78(4), 405–426.

Stohlgren, T. J., Schell, L. D., & Heuvel, B. Vanden. (1999). How Grazing And Soil Quality Affect Native and Exotic Plat Diversity in Rocky Mountain Grassland. Ecological Applications, 9(1), 45–64. https://doi.org/10.1890/1051-0761(1999)009[0045:HGASQA]2.0.CO;2

Stolton, S., Redford, K. H., Dudley, N., Bill, W., Corcuera, E., & Mitchell, B. A. (2014). The Futures of Privately Protected Areas. Gland, Switzerland: IUCN.

Stone, W. W., & Gillion, R. J. (2014). Pesticides in U.S. streams and rivers: occurrence and trends during 1992-2011. Environ Sci technol,48(19), 11025-30. doi: 10.1021/es5025367

Stoner, A. W. 1997. Shell middens as indicators of long-term distributional pattern in Strombus gigas, a heavily exploited marine gastropod. Bulletin of Marine Science 61:559–570.

Stoner, A. W., & Ray, M. (1996). Queen conch, Strombus gigas, in fished and unfished locations of the Bahamas: effects of a marine fishery reserve on adults, juveniles, and larval production. Fishery Bulletin, 94(3), 551-556.

Strecker, U. (2006). The impact of invasive fish on an endemic Cyprinodon species flock (Teleostei) from Laguna Chichancanab, Yucatan, Mexico. Ecology of Freshwater Fish, 15(4), 408–418. https://doi.org/10.1111/j.1600-0633.2006.00159.x

Suárez. A., E. Garraway. D. Vilamajo, L. Mujica, J. Gerhartz, R. Capote, N. Blake 2008. Climate change impacts on terrestrial biodiversity in the insular Caribbean: Report of Working Group III, Climate Change and Biodiversity in the Insular Caribbean. CANARI Technical Report No.383: 99pp.

Sunda, W. G. 2012. Feedback interactions between trace metal nutrients and phytoplankton in the ocean. Frontiers in Microbiology 3:1–22.

Surber, S.J., & Simonton, D.S. (2017). Disparate impacts of coal mining and reclamation concerns for West Virginia and central Appalachia. Resources Policy, 54, 1-8, https://doi.org/10.1016/j.resourpol.2017.08.004

Sutherland, K. P., J. W. Porter, and C. Torres. 2004. Disease and immunity in Caribbean and Indo-Pacific zooxanthellate corals. Marine Ecology Progress Series 266:273–302.

Sutton, M. a, O. Oenema, J. W. Erisman, A. Leip, H. van Grinsven, and W. Winiwarter. 2011. Too much of a good thing. Nature 472:159–161.

Sutton, R., Sedlak, M.D., Yee, D., Davis, J.A., Crane, D., Grace, R. and Arsem, N., 2014. Declines in polybrominated diphenyl ether contamination of San Francisco Bay following production phase-outs and bans. Environmental science & technology, 49(2), pp.777-784.

Swain, E. B., D. R. Engstrom, M. E. Brigham, T. a Henning, and P. L. Brezonik. 1992. Increasing rates of atmospheric mercury deposition in midcontinental north america. Science (New York, N.Y.) 257:784–787.

Swenson, J. J., C. E. Carter, J. C. Domec, and C. I. Delgado. 2011. Gold mining in the peruvian amazon: Global prices, deforestation, and mercury imports. PLoS ONE 6.

Taketani, R. G., N. O. Franco, A. S. Rosado, and J. D. van Elsas. 2010. Microbial community response to a simulated hydrocarbon spill in mangrove sediments. Journal of Microbiology 48:7-15.

Telmer, K.H. and Veiga, M.M., 2009. World emissions of mercury from artisanal and small scale gold mining. In Mercury fate and transport in the global atmosphere (pp. 131-172). Springer US.

Texeira, M., M. Oyarzabal, G. Pineiro, S. Baeza, and J. M. Paruelo. 2015. Land cover and precipitation controls over long-term trends in carbon gains in the grassland biome of South America. Ecosphere 6:art196. <http://www.esajournals.org/doi/10.1890/ES15-00085.1>.

The World Bank Database. (2017): https://data.worldbank.org/indicator/

The World Bank (2017). Terrestrial and Marine protected areas. Available at: http://data.worldbank.org/indicator/ER.PTD.TOTL.ZS?end=2014&page=5&start=2014&view=bar

The World Bank (2016). Global Economic Prospects (2016). Latin America and the Caribbean, Washington

Thomas, M. A. (2010). What Do the Worldwide Governance Indicators Measure? European Journal of Development Research, 22(1), 31–54. http://doi.org/10.1057/ejdr.2009.32

Thompson, J., Charpentier, A., Bouguet, G., Charmasson, F., Roset, S., Buatois, B., Vernet, P., & Gouyon, P.-H. (2013). Evolution of a genetic polymorphism with climate change in a Mediterranean landscape. Proceedings of the National Academy of Sciences of the United States of America, 110(8), 2893–2897. https://doi.org/10.1073/pnas.1215833110

Tijoux, M. E. (2016). Naturaleza americana. Extractivismo y geopolítica del capital. Actuel Marx N° 19: Naturaleza americana. Extractivismo y geopolítica del capital. Actuel Marx N° 19. LOM Ediciones.

Tilman, D., & Clark, M. (2014). Global diets link environmental sustainability and human health. Nature, 515(7528), 518-522.

Tilman, D., C. Balzer, J. Hill, and B. L. Befort. 2011. Global food demand and the sustainable intensification of agriculture. Proc Natl Acad Sci U S A 108:20260–20264. <http://www.ncbi.nlm.nih.gov/pubmed/22106295>.

Tilman, D., Cassman, K. G., Matson, P. A., Naylor, R., & Polasky, S. (2002). Agricultural sustainability and intensive production practices. Nature, 418(6898), 671.

Tlaiye, Laura E.; Aryal, Dinesh. 2013. Latin America and Caribbean Region Environment and Water Resources occasional paper series. Washington DC; World Bank. http://documents.worldbank.org/curated/en/620071468054545005/Ampliando-el-financiamiento-para-la-conservacion-de-la-biodiversidad-las-experiencias-de-America-Latina-y-el-Caribe

Tófoli, R. M., Dias, R. M., Alves, G. H. Z., Hoeinghaus, D. J., Gomes, L. C., Baumgartner, M. T., & Agostinho, A. A. (2017). Gold at what cost? Another megaproject threatens biodiversity in the Amazon. Perspectives in Ecology and Conservation. 15:129-131

Tognetti, P. M., & Chaneton, E. J. (2015). Community disassembly and invasion of remnant native grasslands under fluctuating resource supply. Journal of Applied Ecology, 52(1), 119–128. https://doi.org/10.1111/1365-2664.12349

Tognetti, P. M., Chaneton, E. J., Omacini, M., Trebino, H. J., & León, R. J. C. (2010). Exotic vs. native plant dominance over 20years of old-field succession on set-aside farmland in Argentina. Biological Conservation, 143(11), 2494–2503. https://doi.org/10.1016/j.biocon.2010.06.016

Tovar, C., C. A. Arnillas, F. Cuesta, and W. Buytaert. 2013. Diverging Responses of Tropical Andean Biomes under Future Climate Conditions. PLoS ONE 8.

Townsend, J. M., C. C. Rimmer, C. T. Driscoll, K. P. McFarland, and E. Iñigo-Elias. 2013. Mercury concentrations in tropical resident and migrant songbirds on Hispaniola. Ecotoxicology 22:86–93.

Trace, S., (2016). Rethink, Retool, Reboot: Technology as if people and planet mattered. Rugby, UK: Practical Action Publishing, http://dx.doi.org/10.3362/ 9781780449043

Turtle Conservation Coalition. (2011). Top_25+_Turtles_in_Trouble_2011. U.S. Department of the Interior Fish and Wildlife Service. Notice of Intent to Include Four Native U.S. Freshwater Turtle Species in Appendix III of the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES) 1–35 (2014). http://doi.org/10.1017/CBO9781107415324.004

Twiss, M. R., K. J. Rattan, R. M. Sherrell, and R. M. L. McKay. 2004. Sensitivity of phytoplankton to copper in Lake Superior. J. Great Lakes Res. 30:245–255. <files>.

U.S. Department of the Interior Fish and Wildlife Service. 2014. No Title No Title. Notice of Intent to Include Four Native U.S. Freshwater Turtle Species in Appendix III of the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES).

UN Habitat (2016) Urbanization and development: emerging futures; world cities report 2016. Nairobi, UN Habitat

UNEP 2016. Regional Cooperation for Environmental Sustainability in the Latin American and Caribbean Region. XX Meeting of the Forum of Ministers of the Environment of Latin America and the Caribbean. Cartagena, Colombia.

UNEP, 2004, Global International Waters Assessment Arctic Greenland, East Greenland Shelf, West Greenland Shelf, GIWA Regional assessment 1b, 15, 16.

UNEP. 2010. Latin America and the Caribbean: Environmental outlook. Relations between Environmental Changes and Human WellBeing in Latin America and the Caribbean.

UNEP. 2014. The Importance of Mangroves to People: A Call to Action. J.van Bochove, E. Sullivan, and T. Nakamura, editors. United Nations Environment Programme World Conservation Monitoring Centre, Cambridge. 128 pp.

UNEP.2009. Persistent Organic Pollutants Along Environmental Transects in Costa Rica, Chile, Nepal, and Botswana.

UNESCO. 2016. UNESCO Science Report. Towards 2030. UNESCO Publishing. Second revised edition 2016. Paris.

United Nations Conference on Trade and Development (UNCTAD) (2016). UNCTAD Handbook of Statistics, New York: www.unctad.org

United Nations Development Programme. 2014. Human development report 2014. UN Development Programme. <http://hdr.undp.org/en/media/HDR_2013_EN_complete.pdf>.

United Nations Development Programme. 2016. Human development report 2016. UN Development Programme. http://hdr.undp.org/sites/default/files/2016_human_development_report.pdf

United Nations, (2016). Report of the Special Rapporteur on the rights of indigenous peoples on her visit to Honduras. Human Rights Council. Thirty-third session. Agenda item 3. Promotion and protection of all human rights, civil, political, economic, social and cultural rights, including the right to development 21 July 2016. United Nations , A/HRC/33/42/Add.2

United Nations, Department of Economic and Social Affairs, Population Division (2014). World Urbanization Prospects: The 2014 Revision, Highlights (ST/ESA/SER.A/352).

Uribe B., E., (2015). Estudios del cambio climático en América Latina: El cambio climático y sus efectos en la biodiversidad en América Latina. CEPAL. https://repositorio.cepal.org/handle/11362/39855

USGS 2016 http://landcovertrends.usgs.gov/east/eco63Report.html

Valdez-Moreno, M., Quintal-Lizama, C., Gómez-Lozano, R., & García-Rivas, M. del C. (2012). Monitoring an alien invasion: DNA barcoding and the identification of lionfish and their prey on coral reefs of the Mexican Caribbean. PLoS ONE, 7(6), 1–8. https://doi.org/10.1371/journal.pone.0036636

Vales, M., Álvarez, A., Montes, L., & Ávila A. (Eds.). (1998). Estudio Nacional sobre la Diversidad Biológica en la República de Cuba. CESYTA, Madrid, Spain.

Valiela, I., J. L. Bowen, and J. York. 2001. Mangrove Forests: one of the world's threatened major tropical environments. BioScience 51: 807-815.

Valliere, J. M., Irvine, I. C., Santiago, L., & Allen, E. B. (2017). High N, dry: Experimental nitrogen deposition exacerbates native shrub loss and nonnative plant invasion during extreme drought. Global Change Biology.

Van Aardenne, J., Dentener, F.J., Olivier, J.G.J., Goldewijk, C.G.M. and Lelieveld, J., 2001. A 1× 1 resolution data set of historical anthropogenic trace gas emissions for the period 1890–1990. Global Biogeochemical Cycles, 15(4), pp.909-928.

van Andel, T., van der Hoorn, B., Stech, M., Arostegui, S. B., & Miller, J. (2016). Global Ecology and Conservation.

Van Beusekom, A. E., G. Gonz??lez, and M. M. Rivera. 2015. Short-term precipitation and temperature trends along an elevation gradient in northeastern Puerto Rico. Earth Interactions 19:3–33.

van der Burg, W., Freitas, J. De, & Debrot, A. O. (2012). Naturalised and invasive alien plant species in the Caribbean Netherlands : status, distribution, threats, priorities and recommendations. Dutch Minsitry of Economic Affairs, (October).

van Hooidonk, R., J. A. Maynard, Y. Liu, and S.-K. Lee, 2015: Downscaled projections of Caribbean coral bleaching that can inform conservation planning. Global Change Biol., 21: 3389–3401, doi:10.1111/gcb.12901.

Van Kleunen, M., W. Dawson, F. Essl, J. Pergl, M. Winter, E. Weber, H. Kreft, P. Weigelt, J. Kartesz, M. Nishino, L. A. Antonova, J. F. Barcelona, F. J. Cabezas, O. Morozova, D. Moser, D. L. Nickrent, A. Patzelt, P. B. Pelser, M. P. Baptiste, M. Poopath, M. Schulze, H. Seebens, W.-S. Shu, J. Thomas, M. Velayos, and J. J. Wieringa. 2015. Global exchange and accumulation of non-native plants. Nature 525:100–103.

Van Lexmond, M.B., Bonmatin, J.M., Goulson, D. and Noome, D.A., 2015. Worldwide integrated assessment on systemic pesticides.

Van Ruijven, B.J., K. Daenzer, K. Fisher-Vanden, T. Kober, S. Paltsev, R. H. Beach, S. L. Calderon, K. Calvin, M. Labriet, A. Kitous, A.F.P. Lucena, D. P. van Vuurenk. (2016), Baseline projections for Latin America: base-year assumptions, key drivers and greenhouse emissions. Energy Economics, 56: 498-511.

Van Wagner, V. (1978). Age-class distribution and the forest fire cycle. Canadian Journal of Forest Research, 8(2), 220-227.

Vanhulst, J., & Beling, A. E. (2014). Buen vivir: Emergent discourse within or beyond sustainable development? Ecological Economics, 101, 54–63. http://doi.org/10.1016/j.ecolecon.2014.02.017

Vaquer-Sunyer, R., & Duarte, C.M. (2008). Thresholds of hypoxia for marine biodiversity. Proceedings of the National Academy of Sciences, 105, 15452-15457, 10.1073/pnas.0803833105

Vargas, C.A., Aguilera, V.M., San Martín, V., Manríquez, P.H., Navarro, J.M., Duarte, C., Torres, R., Lardies, M.A. and Lagos, N.A., 2015. CO2-driven ocean acidification disrupts the filter feeding behavior in Chilean gastropod and bivalve species from different geographic localities. Estuaries and coasts, 38(4), pp.1163-1177.

Vargas, C.A., Contreras, P.Y., Pérez, C.A., Sobarzo, M., Saldías, G.S. and Salisbury, J., 2016. Influences of riverine and upwelling waters on the coastal carbonate system off Central Chile and their ocean acidification implications. Journal of Geophysical Research: Biogeosciences, 121(6), pp.1468-1483.

Vega-Rodriguez, M., Müller-Karger, F., Hallock, P., Quiles-Perez, G., Eakin, C., Colella, M., Jones, D., Li, J., Soto, I., Guild, L., Lynds, S., & Ruzicka, R. (2015). Influence of water-temperature variability on stony coral diversity in Florida Keys patch reefs. Marine Ecology Progress Series, 528, 173–186. https://doi.org/10.3354/meps11268

Veldman, J. W., & Putz, F. E. (2010). Long-distance dispersal of invasive grasses by logging vehicles in a tropical dry forest. Biotropica, 42(6), 697–703. https://doi.org/10.1111/j.1744-7429.2010.00647.x

Vellend, M., Harmon, L. J., Lockwood, J. L., Mayfield, M. M., Hughes, A. R., Wares, J. P., & Sax, D. F. (2007). Effects of exotic species on evolutionary diversification. Trends in Ecology & Evolution, 22(9), 481-488.

Venier, M. and Hites, R.A., 2010. Time trend analysis of atmospheric POPs concentrations in the Great Lakes region since 1990. Environmental science & technology, 44(21), pp.8050-8055.

Venturini, N., M. C. Bicego, S. Taniguchi, S. T. Sasaki, F. Garcia-Rodriguez, E. Brugnoli, and P. Muniz. 2015. A multi-molecular marker assessment of organic pollution in shore sediments from the Rio de la Plata Estuary, SW Atlantic. Marine Pollution Bulletin 91:461–475.

Veuthey, Sandra, and Julien-Francois Gerber. "Accumulation by dispossession in coastal Ecuador: Shrimp farming, local resistance and the gender structure of mobilizations." Global Environmental Change 22.3 (2012): 611-622.

Victoria, R. L., L. a Martinelli, M. Moraes, M. V Ballester, a. V Krusche, G. Pellegrino, R. M. B. Almeida, and J. E. Richey. 1998. Surface Air Temperature Variations in the Amazon Region and Its Borders during This Century. Journal of Climate 11:1105–1110.

Vignola, R., Locatelli, B., Martinez, C., & Imbach, P. (2009). Ecosystem-based adaptation to climate change: what role for policy-makers, society and scientists?. Mitigation and adaptation strategies for global change, 14(8), 691.

Vilà, M., & Ibáñez, I. (2011). Plant invasions in the landscape. Landscape Ecology, 26(4), 461–472.

Villalba-Eguiluz, C. U., & Etxano, I. (2017). Buen Vivir vs Development (II): The Limits of (Neo)Extractivism. Ecological Economics, 138, 1–11. http://doi.org/10.1016/j.ecolecon.2017.03.010

Villarini, G., Smith, J. A., Baeck, M. L., Vitolo, R., Stephenson, D. B., & Krajewski, W. F. (2011). On the frequency of heavy rainfall for the Midwest of the United States. Journal of Hydrology, 400(1-2), 103-120.

Villers R.L. & Hernández LJ. 2007. Incendios forestales y el fenómeno de El Niño en México. Resumenes de las comunicaciones de la IV Conferencia Internacional sobre incendios forestales. 13-17 mayo de 2007, Sevilla España.

Vincent, L. A., T. C. Peterson, V. R. Barros, M. B. Marino, M. Rusticucci, G. Carrasco, E. Ramirez, L. M. Alves, T. Ambrizzi, M. A. Berlato, A. M. Grimm, J. A. Marengo, L. Molion, D. F. Moncunill, E. Rebello, Y. M. T. Anunciação, J. Quintana, J. L. Santos, J. Baez, G. Coronel, J. Garcia, I. Trebejo, M. Bidegain, M. R. Haylock, and D. Karoly. 2005. Observed trends in indices of daily temperature extremes in South America 1960-2000. Journal of Climate 18:5011–5023.

Virtanen, J.K., Voutilainen, S., Rissanen, T.H., Mursu, J., Tuomainen, T.P., Korhonen, M.J., Valkonen, V.P., Seppänen, K., Laukkanen, J.A. and Salonen, J.T., 2005. Mercury, fish oils, and risk of acute coronary events and cardiovascular disease, coronary heart disease, and all-cause mortality in men in eastern Finland. Arteriosclerosis, thrombosis, and vascular biology, 25(1), pp.228-233.

Višnjevec, A.M., Kocman, D. and Horvat, M., 2014. Human mercury exposure and effects in Europe. Environmental toxicology and chemistry, 33(6), pp.1259-1270.

Vogt, N. D., M. Pinedo-Vasquez, E. S. Brondízio, O. Almeida, and S. Rivero. 2015. Forest Transitions in Mosaic Landscapes: Smallholder's Flexibility in Land-Resource Use Decisions and Livelihood Strategies From World War II to the Present in the Amazon Estuary. Society & Natural Resources 28:1043–1058.

Volante, J., Mosciaro, J., Morales Poclava, M., Vale, L., Castrillo, S., Sawchik, J., Tiscornia, G., Fuente, M., Maldonado, I., Vega, A., Trujillo, R., CortéZ, L., & Paruelo, J. (2015). Expansión agrícola en Argentina, Bolivia, Paraguay, Uruguay y Chile entre 2000-2010. Caracterización espacial mediante series temporales de índices de vegetación. Revista de Investigaciones Agripecurias, 41(2), 179–191. Retrieved from http://ppct.caicyt.gov.ar/index.php/ria/article/view/4704/6286

Volk, M., P. Bungener, F. Contat, M. Montani, and J. Fuhrer. 2006. Grassland yield declined by a quarter in 5 years of free-air ozone fumigation. Global Change Biology 12:74–83.

Vos, Vincent A.; Olver Vaca y Adrián Cruz (2015). "Sistemas Agroforestales en la Amazonía boliviana. Una valoración de sus múltiples funciones", Cuadernos de Investigación 82, La Paz, junio de 2015.

Vranken, L., T. Avermaete, D. Petalios and E. Mathijs (2014). "Curbing global meat consumption: Emerging evidence of a second nutrition transition." Environmental Science & Policy 39: 95-106.

Vuille, M., E. Franquist, R. Garreaud, W. Sven, L. Casimiro, and B. Cáceres. 2015. Journal of Geophysical Research: Atmospheres.

Vuille, M., R. S. Bradley, M. Werner, and F. Keimig. 2003. 20Th Century Climate Change in the Tropical Andes: Observations and Model Results. Climatic Change 59:75–99.

Wallem, P.K., C.B. Anderson, G. Martínez Pastur & M.V. Lencinas (2010). Community re-assembly by an exotic herbivore,Castor canadensis, in subantarctic forests, Chile and Argentina. Biological Invasions 12: 325-335. Special edition on invasive herbivores.

Walters, W. (2016). Border/control. In An Anthology of Migration and Social Transformation (pp. 151-165). Springer International Publishing.

Ward, J. M., & Ricciardi, A. (2007). Impacts of Dreissena invasions on benthic macroinvertebrate communities: A meta-analysis: Biodiversity research. Diversity and Distributions, 13(2), 155–165. https://doi.org/10.1111/j.1472-4642.2007.00336.x

Ward, R. D., D. A. Friess, R. H. Day, and R. A. MacKenzie. 2016. Impacts of climate change on mangrove ecosystems: a region by region overview. Ecosystem Health and Sustainability 2:e01211.

Wardle D.A., Bardgett R.D., Callaway R.M. & Van Der Putten W.H. (2011). Terrestrial ecosystem responses to species gains and losses. Science, 332, 1273-1277

Washington, S., & Ababouch, L. (2011). Private standards and certification in fisheries and aquaculture: current practice and emerging issues. FAO Fisheries and Aquaculture Technical Paper. http://doi.org/10.1017/CBO9781107415324.004

Weber, B., Belnap, J., & Büdel, B. (2016). Synthesis on Biological Soil Crust Research. In Biological Soil Crusts: An Organizing Principle in Drylands (pp. 527-534). Springer International Publishing.

Webster, N. S., S. Uthicke, E. S. Botté, F. Flores, and A. P. Negri. 2013. Ocean acidification reduces induction of coral settlement by crustose coralline algae. Global Change Biology 19:303–315.

Weihe, P., Hansen, J.C., Murata, K., Debes, F., Jørgensen, P.J., Steuerwald, U., White, R.F. and Grandjean, P., 2002. Neurobehavioral performance of Inuit children with increased prenatal exposure to methylmercury. International Journal of Circumpolar Health, 61(1), pp.41-49.

Weinhold, D., Killick, E., & Reis, E. J. (2013). Soybeans, poverty and inequality in the Brazilian Amazon. World Development, 52, 132-143.

Weiser, E. L., & Powell, A. N. (2010). Does garbage in the diet improve reproductive output of Glaucous Gulls?. The Condor, 112(3), 530-538.

Weiss-Penzias, P. S., D. A. Gay, M. E. Brigham, M. T. Parsons, M. S. Gustin, and A. ter Schure. 2015. Trends in mercury wet deposition and mercury air concentrations across the U.S. and Canada. Science of the Total Environment 1–11.

Weiss, J. L., and J. T. Overpeck. 2005. Is the Sonoran Desert losing its cool? Global Change Biology 11:2065–2077.

Welcomme, R. L. 1999. A review of a model for qualitative evaluation of exploitation levels in multi-species fisheries. Fisheries Management and Ecology 6:1–19.

Welcomme, R. L., I. G. Cowx, D. Coates, C. Béné, S. Funge-Smith, A. Halls, and K. Lorenzen. 2011. Inland capture fisheries.

Welle T, Birkmann J, Rhyner J, Witting M, Wolfertz J (2012) WorldRiskIndex 2012: concept and results. In Alliance Development Works (eds.): WorldRiskReport. Berlin, pp. 11–26

Wertin, T. M., Reed, S. C., & Belnap, J. (2015). C3 and C4 plant responses to increased temperatures and altered monsoonal precipitation in a cool desert on the Colorado Plateau, USA. Oecologia, 177(4), 997-1013.

Whitfield, P. E., Gardner, T., Vives, S. P., Gilligan, M. R., Courtenay Jr, W. R., Ray, G. C., & Hare, J. A. (2002). Biological invasion of the Indo-Pacific lionfish Pterois volitans along the Atlantic coast of North America. Marine Ecology Progress Series, 235, 289-297.

Wiebusch & Lant 2017. Policy Drivers of US Wetland Conversion Rates, 1955-2009, Society and Natural Resources 30:16-30.

Wiedenfeld, D.A., Crawford, R. and Pott, C.M., 2015. Workshop Report: Reducing the Bycatch of Seabirds, Sea Turtles, and Marine Mammals in Gillnets. 21-23 January 2015. American Bird Conservancy and BirdLife International.

Wik, M., Pingali, P., & Brocai, S. (2008). Global agricultural performance: past trends and future prospects.

Wilcove, D. S., D. Rothstein, J. Dubow, A. Phillips, and E. Losos. 1998. Quantifying threats to imperiled species in the United States. BioScience 48:607–615.

Wilcox, C., N.J. Mallos, G. H. Leonard, A. Rodriguez, B. D. Hardesty, 2016, Using expert elicitation to estimate the impacts of plastic pollution on marine wildlife. Marine Policy 65:107–114.

Wilkie, D.S. and Godoy, R.A. (2001) 'Income and price elasticities of Bushmeat Demand Lowland Amerindian Societies', Conservation Biology,153, 761-769.

Wilkinson, C. R., & Souter, D. N. (Eds.). (2008). Status of Caribbean coral reefs after bleaching and hurricanes in 2005 (Vol. 148). Global Coral Reef Monitoring Network.

Willer, H. and Lernoud, J., 2016. The world of organic agriculture. Statistics and emerging trends 2016 (pp. 1-336). Research Institute of Organic Agriculture FiBL and IFOAM Organics International.

Wilsey, B. J., Martin, L. M., & Polley, H. W. (2005). Predicting Plant Extinction Based on Species-Area Curves in Prairie Fragments with High Beta Richness. Conservation Biology, 19(6), 1835-1841.

Wirtz, D., Sorensen, D. C., & Haasdonk, B. (2014). A posteriori error estimation for DEIM reduced nonlinear dynamical systems. SIAM Journal on Scientific Computing, 36(2), A311-A338.

Wittmann, F., Schongart, J., Montero, J. C., Motzer, T., Junk, W. J., Piedade, M. T. F., Queiroz, H. L., & Worbes, M. (2006). Tree species composition and diversity gradients in white-water forests across the Amazon Basin. Journal of Biogeography, 33(8), 1334–1347. https://doi.org/10.1111/j.1365-2699.2006.01495.x

Wolfe, B. T., & Van Bloem, S. J. (2012). Subtropical dry forest regeneration in grass-invaded areas of Puerto Rico: Understanding why Leucaena leucocephala dominates and native species fail. Forest Ecology and Management, 267, 253–261. https://doi.org/10.1016/j.foreco.2011.12.015

Wolfe, S.A., D.B. Griffith, and C.A. Gray Wolfe. 2000. Response of reindeer and caribou to human activities. Polar Research 19:63-73.

Woodley, S., Bertzky, B., Crawhall, N., Dudley, N., Londoño, J. M., Mackinnon, K., & Redford, K. (2012). Meeting aichi target 11: what does success look like for protected area systems? Parks, 18(1).

Wooldridge, S.A. and Done, T.J., 2009. Improved water quality can ameliorate effects of climate change on corals. Ecological Applications, 19(6), pp.1492-1499.

Woodruff, T.J., Grillo, J. and Schoendorf, K.C., 1997. The relationship between selected causes of postneonatal infant mortality and particulate air pollution in the United States. Environmental health perspectives, 105(6), p.608.

Woody, C. A., Hughes, R. M., Wagner, E. J., Quinn, T. P., Roulson, L. H., Martin, L. M., & Griswold, K. (2010). The Mining Law of 1872: Change is Overdue. Fisheries, 35(7), 321–331. https://doi.org/10.1577/1548-8446-35.7.321

Worldometers (2017). Accessed 2 May 2017, and 3 September 2017 at: http://www.worldometers.info/world-population/population-by-region/

Worm, B., H. K. Lotze, I. Jubinville, C. Wilcox, J. Jambeck, 2017, Plastic As a Persistent Marine Pollutant, Annual Review of Environment and Resources, 42:1

Wortley, L., Hero, J. M., & Howes, M. (2013). Evaluating ecological restoration success: a review of the literature. Restoration Ecology, 21(5), 537-543.

Wright, C. K., & Wimberly, M. C. (2013). Recent land use change in the Western Corn Belt threatens grasslands and wetlands. Proceedings of the National Academy of Sciences, 110(10), 4134-4139.

Wu, Y., Wang, S., Streets, D.G., Hao, J., Chan, M. and Jiang, J., 2006. Trends in anthropogenic mercury emissions in China from 1995 to 2003. Environmental science & technology, 40(17), pp.5312-5318.

Wurster, C.F. and Wingate, D.B., 1968. DDT residues and declining reproduction in the Bermuda petrel. Science, 159(3818), pp.979-981.

Wurster, D.H., Wurster, C.F. and Strickland, W.N., 1965. Bird mortality following DDT spray for Dutch elm disease. Ecology, 46(4), pp.488-499.

WWF. (2014). Living Planet Report 2014: Species and spaces, people and places. Research accunting 1:180. <http://www.ncbi.nlm.nih.gov/pubmed/15003161\nhttp://cid.oxfordjournals.org/lookup/doi/10.1093/cid/cir991\nhttp://www.scielo.cl/pdf/udecada/v15n26/art06.pdf\nhttp://www.scopus.com/inward/record.url?eid=2-s2.0-84861150233&partnerID=tZOtx3y1>.

WWF. (2016). Living Planet Report 2016: Risk and resilience in a new era: https://wwf.panda.org/knowledge_hub/all_publications/lpr_2016/. Date accessed: March 20th, 2017.

Yang, Z., T. Wang, R. Leung, K. Hibbard, T. Janetos, I. Kraucunas, J. Rice, B. Preston, and T. Wilbanks. 2014. A modeling study of coastal inundation induced by storm surge, sea-level rise, and subsidence in the Gulf of Mexico. Natural Hazards 71:1771–1794.

Yearbook, FAO Statistical. "World food and agriculture." Food and Agriculture Organization of the United Nations, Rome (2013): 2013.

Yee, S. H., D. L. Santavy, and M. G. Barron. 2008. Comparing environmental influences on coral bleaching across and within species using clustered binomial regression. Ecological Modelling 218:162–174.

Zagarola, J.P., C.B. Anderson & J. Veteto (2014). Perceiving Patagonia - An assessment of social values and perspectives regarding watershed ecosystem services and management in southern South America.Environmental Management 53: 769-782.

Zambrano, L., Valiente, E., & van der Zanden, M. J. (2010). Food web overlap among native axolotl (Ambystoma mexicanum) and two exotic fishes: Carp (Cyprinus carpio) and tilapia (Oreochromis niloticus) in Xochimilco, Mexico City. Biological Invasions, 12(9), 3061–3069. https://doi.org/10.1007/s10530-010-9697-8

Zenni, R. D., & Ziller, S. R. (2011). An overview of invasive plants in Brazil. Revista Brasileira de Botânica, 34(3), 431–446. https://doi.org/10.1590/S0100-84042011000300016

Zenni, R. D., Ziller, S. R., Pauchard, A., Rodriguez-Cabal, M., & Nuñez, M. A. (2017). Invasion science in the developing world: A response to Ricciardi et al. Trends in Ecology & Evolution, 32(11), 807-808.

Zéphyr, P. M. D., Guillén, A. C., Salgado, H., & Seligson, M. A. (2011). Haiti in distress: The impact of the 2010 earthquake on citizen lives and perceptions. LAPOP.

Zhou, Y., A. M. Michalak, D. Beletsky, Y. R. Rao, and R. P. Richards. 2015. Record-breaking Lake Erie hypoxia during 2012 drought. Environmental Science and Technology 49:800–807.

Zilberman, D., Hochman, G., Rajagopal, D., Sexton, S., & Timilsina, G. (2012). The impact of biofuels on commodity food prices: Assessment of findings. American Journal of Agricultural Economics, 95(2), 275-281.

Ziller, S. R., Reaser, J. K., Neville, L. E., & Brand, K. (2005). Invasive alien species in South America: national reports & directory of resources. Prevention and Management of Invasive Alien Species: Forging Cooperation throughout South America, 114 p. Retrieved from www.gisp.org

Zimmerle, D.J., Williams, L.L., Vaughn, T.L., Quinn, C., Subramanian, R., Duggan, G.P., Willson, B., Opsomer, J.D., Marchese, A.J., Martinez, D.M. and Robinson, A.L., 2015. Methane emissions from the natural gas transmission and storage system in the United States. Environmental science & technology, 49(15), pp.9374-9383.

Zulkafli, Z., W. Buytaert, B. Manz, C. V. Rosas, P. Willems, W. Lavado-Casimiro, J.-L. Guyot, and W. Santini. 2016. Projected increases in the annual flood pulse of the Western Amazon. Environmental Research Letters 11:014013.

5 Chapter 5: Current and future interactions between nature and society

Coordinating Lead Authors
Brian J. Klatt (USA), Jaime Ricardo García Márquez (Colombia/Germany), Jean Pierre Ometto (Brazil)

Lead Authors
María Piedad Baptiste Espinosa (Colombia), Sara Wilson S. (Canada), Sandra Verónica Acebey Quiroga (Bolivia), María Claudia Guezala Villavicencio (Peru), Matias Enrique Mastrangelo (Argentina), Walter Alberto Pengue (Argentina), Mariela Verónica Blanco (Argentina), Tatiana Gadda (Brazil), Wilson Ramírez Hernández (Colombia), John Agard (Trinidad and Tobago).

Fellow
Mireia Valle (Spain/Ecuador)

Contributing Authors
Marcello Hernandez-Blanco (Costa Rica), Brenda Parlee (Canada)

Review Editors
Ana Carolina Carnaval (Brazil/USA), Peter Kareiva (USA)

To be cited as:
Klatt, B. J., García Márquez., J. R., Ometto, J. P., Valle, M., Mastrangelo, M. E., Gadda, T., Pengue, W. A., Ramírez Hernández., W., Baptiste Espinosa., M. P., Acebey Quiroga., S.V., Blanco, M. V., Agard, J., and Guezala Villavicencio., M. C. Chapter 5: Current and future interactions between nature and society. In IPBES (2018): The IPBES regional assessment report on biodiversity and ecosystem services for the Americas. Rice, J., Seixas, C. S., Zaccagnini, M. E., Bedoya-Gaitán, M., and Valderrama, N. (eds.). Secretariat of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services, Bonn, Germany, pp. xx-xx

Table of contents Chapter 5

5.1 Executive summary

1. **One hundred per cent of the natural units of analysis will continue to be negatively affected, with a concomitant decrease in nature's contributions to people, given current trends (business as usual), though the magnitude and exact mechanism of the individual drivers will vary by driver and unit of analysis** (e*stablished but incomplete)*{5.4}. For example, tropical moist and dry forest and coastal mangroves will continue to exhibit a decline due to land use change regardless of the scenarios considered, but different local factors (agriculturalization and urbanization, respectively) will be involved *(well established)* {5.4.1, 5.4.11}. Additionally, some drivers will affect units of analysis differently. Empirical evidence indicates differential effects of climate change: boreal forest is extending northward {5.4.2}, while tundra is diminishing in land area *(established but incomplete)* {5.4.3}*.* Thus, some drivers, and their relative roles, will need to be further refined on a local scale and with respect to their proximate factors.

2. **Multiple drivers will act in synergy and further produce biodiversity loss and impact nature's contributions to people in most of the units of analysis for the Americas** *(established but incomplete*){5.4}. Climate change, combined with other drivers, is predicted to account for an increasingly larger proportion of biodiversity loss in the future, in both terrestrial and aquatic ecosystems {5.3}. Forest fragmentation, climate change and industrial development increase risk of biodiversity and nature's contributions to people loss i.e. dry forest unit of analysis {5.4.1.2}. Predictions on invasive species and climate change indicates an increase in habitable areas and their potential impacts on different units of analysis {5.3}.

3. **Changes in temperature, precipitation regime and extreme climate events are predicted to impact all units of analysis in the Americas** *(well established)* {5.4}. Climate change and the potential impacts on tropical dry forests by changing the frequency of wildfires; change in forest structure and functional composition in the Amazon tropical moist forest; extreme drought events changing nature's contributions to people in the Amazon region; insect outbreaks and changes in albedo are predicted to significantly impact temperate, boreal and tundra units of analysis, affecting society and indigenous communities and well-being {5.4}.

4. **Thresholds, or tipping points (conditions resulting in rapid and potentially irreversible changes) may have already been exceeded for some ecosystems and are likely for others** *(established but incomplete).* For instance, it is considered more likely than not that such a threshold has already been passed in the cryosphere with respect to summer sea ice (*established but incomplete*) {5.4.12}. Model simulations indicate changes in forest structure and species distribution in the Amazon forest in response to global warming and change in precipitation patterns (forest die-back) *(established but incomplete)* {5.4.1}. So too, a 4ºC increase in global temperatures is predicted to likely cause widespread die off of boreal forest due to greater susceptibility to disease {5.4.2} and global temperature increases may have already started persistent thawing of the permafrost {5.4.3}. Under 4°C warming, widespread coral reef mortality is expected with significant impacts on coral reef ecosystems {5.4.11}. Sea surface water temperature increase will cause a reduction of sea grass climatic niche: those populations under seawater surface temperature thresholds higher than the temperature ranges required by the species could become extinct by 2100 with concomitant loss of ecosystem services.

5. **Changes in nature and nature's contributions to people in most units of analysis are increasingly driven by causal interactions between distant places (i.e. telecouplings) *(well established)* {5.6.3}, thus scenarios and models that incorporate telecouplings will better inform future policy decisions.** Nature and nature's contributions to people in telecoupled systems can be affected negatively or positively by distant causal interactions. Provision of food and medicine from wild organisms in temperate and tropical grasslands, savannas and forests of South America is being dramatically reduced due to land-use changes driven by the demand of agricultural commodities (e.g. soybeans) mainly from Europe and China. Conservation of insectivorous migratory bats in Mexico benefits pest control in agroecosystems of North America, resulting in increased yields and reduced pesticide costs. Trade policies and international agreements will thus have an increasingly strong effect on environmental outcomes in telecoupled systems.

6. **Policy interventions have resulted in significant land use changes at the local and regional scales and will continue to do so through 2050. These policies have affected nature's contributions to people both positively and negatively, and provide an opportunity to manage trade-offs among nature's contributions to people *(well established)*** {5.4}. Land use changes are now mainly driven by high crop demand, big hydropower plans, rapid urban growth and result in a continued loss of grasslands {5.4.4, 5.4.5}. However, strategies for establishing conservation units have helped in reducing deforestation in the Brazilian Amazon from the period of 2004 to 2011 (*well established*) {5.4.1}. Similarly, wetland protection policies and regulation have helped reduce the conversion of wetlands in North America {5.4.7}. Policies based on command and control measures may be limited in providing effective reduction in ecosystem loss and should be complemented with policies acknowledging multiple values {5.6.3}.

7. **Policy interventions at vastly differing scales (from national to local) lead to successful outcomes in mitigating impacts to biodiversity** *(established but incomplete)*{5.4}. For instance, long-established governmental protections of wetlands in North America have significantly slowed and may have stopped wetland loss based on acreage {5.4.7}. In South America, where mangrove loss continues at a rate of one to two per cent, different stakeholders such as local communities and/or governments have been successful in protecting mangroves based on empowerment and shared interests in their preservation {5.4.11}.

8. **Pressures to nature are projected to increase by 2050, negatively affecting biodiversity as indicated by a potential reduction of the mean species abundance index. However, the magnitude of the pressures by 2050 are expected to be less under transition pathways to sustainability in comparison to the business as usual scenario (*established but incomplete*), {5.5}.** The Global Biodiversity model projected that under the business as usual scenario mean species abundance had decreased in the Americas by approximately 30 per cent by 2010 compared to its values prior to European settlement of the New World, with historical losses primarily attributed to land transformation to agricultural uses. Using the Global Biodiversity model, there is an additional projected loss of 9.6 per cent by 2050, primarily attributed to some additional land use changes , and especially to climate change, which will steadily increase relative to other drivers considered in the model. However, under the transition pathways to sustainability of global technologies, decentralised solutions, and consumption change pathways, the projected losses are 6 per cent, 5 per cent, and 5 per cent, respectively,

achieving a relative improvement of approximately 30 per cent to 50 per cent compared to the business as usual scenario. Under these pathways, climate change mitigation, the expansion of protected areas and the recovery of abandoned lands would significantly contribute to reducing biodiversity loss.

9. **Participative scenarios have proven to be a successful tool for envisioning potential futures and pathways and to embrace and integrate multiple and sometime conflicting values and their role in promoting bottom-up decision making in the face of future's uncertainties (*well established*)** {5.3}**.** The use of participative approaches to develop scenarios has increased during recent years in the Americas. The inclusion of different stakeholders and their knowledges in the process of constructing potential futures has promoted a better understanding of the complexity of the social-ecological systems in which they are embedded. This has enhanced co-learning processes between all actors involved, even those normally under-represented in decision-making activities. As a result, several participative scenario exercises have motivated community-based solutions and local governance initiatives all pointing towards the development of adaptive management strategies {5.3}.

10. **Pathways that consider changes in societal options will lead to less pressure to nature (*established but incomplete*)** {5.6.3}. An example is the indirect impact that shifts in urban dietary preferences have on agricultural production and expansion, and food options that are expected to continue growing into the future. Therefore, not only is there a strong connection between urbanization and economic growth, but also between affluence (and urban preferences) and the global displacement of land use particularly from high-income to low-income countries.

11. **Available local studies informing regional futures of nature and nature's benefit to people do not allow scalability as of yet (*well established*)** {5.3}. The challenge in expanding the findings from local studies resides in the fact that a number of comparable local studies are still not available. Information is scattered throughout the region by the use of different units, methods and scales, which prevents a local-to-regional generalization. The list of "nature" indicators used in studies at local scales is large and heterogeneous (*well established*). Even for the same indicator (e.g. biodiversity), different metrics are used (e.g. species-area curve, mean species abundance) {5.5}. In other cases, multiple indicators are used to describe different aspects of biodiversity and ecosystem services. In this latter case, synergies and trade-offs are explicitly mentioned with a clear pattern in which increasing the provision of some indicators result in the detriment of others {5.3}. For example, agriculture expansion leading to loss in biodiversity illustrates a common trend from local studies expected to continue into the future.

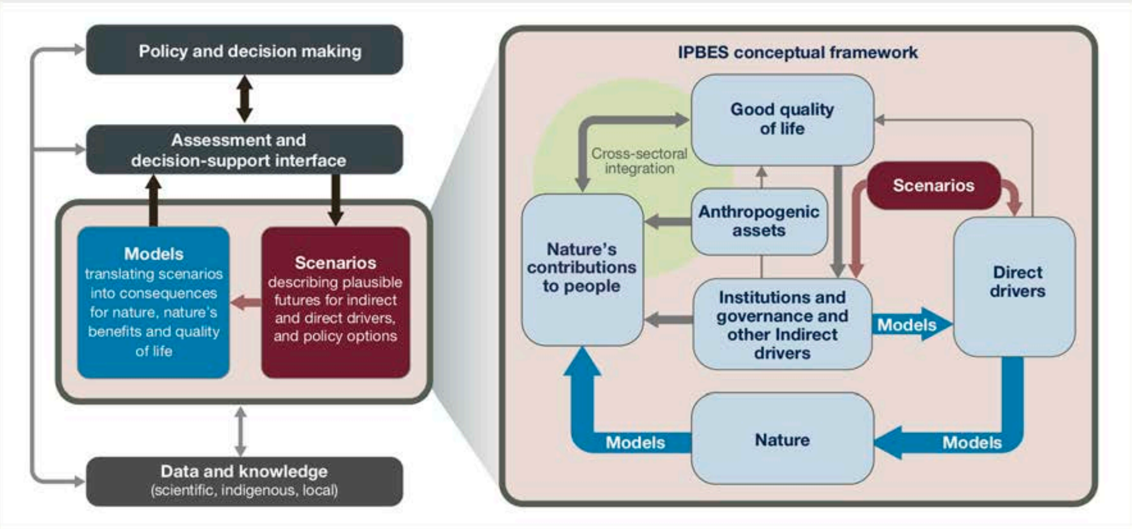
12. **There is a significant research gap in the development of models and scenarios that integrate drivers, nature, nature's contributions to people and good quality of life (*well established*)**{5.3}. Models and scenarios can be powerful tools to integrate and synthesize the complex dynamics of coupled human and nature systems, and to project their plausible behaviors into the future. Most existing models and scenarios focus on the link between drivers and its impacts on nature. Few cases exist in which models or scenarios integrate the relationships between changes in nature and changes in nature's contributions to people and good quality of life {5.3}. Inter-and trans-disciplinary modeling efforts will be required to address this research gap {5.3}.

5.2 Introduction

The IPBES (Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services) conceptual framework illustrates the complex relationships between natural systems and human well-being and how these relationships are determined through the interdependence of the various components. These components include the specific biological system, Nature's Contributions to People (NCP, which includes both ecosystem goods and services and natures' gifts), direct and indirect drivers affecting the system, and the perceived value of the NCP. Previous chapters considered the breadth of NCP, the status and trends of biodiversity, and the major direct and indirect drivers affecting NCP. This chapter aims to: 1) integrate these components by examining what is known with respect to the relationships between them in the Americas; 2) examine what the future state of biodiversity and NCP may be under different plausible future conditions (i.e. "scenarios"); and 3) discuss the establishment of a framework, or pathway, to inform the policy process to attain a sustainable future.

To achieve integration of the framework components we relied on two sources of information: 1) the empirical information presented in earlier chapters of this assessment; and 2) modeling studies. As described in 1.2.6, (IPBES, 2016), and as depicted in Figure 5.1, models are "qualitative or quantitative descriptions of key components of a system and of the relationships between those components", which can be used to assess how systems function or how changes in a system may result in altered outcomes. In the case of this chapter, models involving the components of the IPBES framework can inform us as to likely future conditions, the possible result of policy interventions, or help us define pathways to a more sustainable future and more equitable distribution of NCP among sectors of society or regions. However, it should be noted that even the best models are only approximations of reality and they all have some degree of uncertainty associated with them (Maier et al., 2016). We then evaluated this information through the lens of four major classes of scenarios.

Figure 5 1 IPBES Conceptual framework and high-level roles of scenarios and models in assessment and decision support. Source: IPBES (2016).



Due to the complexity of the issue of biodiversity and NCP, as well as the universe of possible policy interventions, there are an almost infinite number of scenarios that can be constructed and on which models can be based; Hunt et al. (2012) report that over 450 scenarios relating to NCP have been developed. However, as compellingly argued by Hunt et al. (2012), van Vuuren et al. (2012), IPBES (2016) and Kubiszewski et al. (2017), scenarios can be grouped according to a limited number of "archetypes" or families, originally identified by the Global Scenario Group (Gallopin & Rijsberman, 1997). The archetypes encompass four main themes: 1) Market Forces; 2) Fortress World; 3) Policy Reform; and 4) Great Transition.

- Market Forces: This scenario is a story of a market-driven world in the 21st century in which demographic, economic, environmental, and technological trends unfold without major surprises.
- Policy Reform: This scenario envisions the emergence of strong political will for taking harmonized and rapid action to ensure a successful transition to a more equitable and environmentally resilient future.
- Fortress World: This scenario is a variant of a broader class of Barbarization scenarios in the hierarchy of the Global Scenario Group. Barbarization scenarios envision the grim possibility that the social, economic and moral underpinnings of civilization deteriorate, as emerging problems overwhelm the coping capacity of both markets and policy reforms.
- Great Transition: This scenario explores visionary solutions to the sustainability challenge, including new socioeconomic arrangements and fundamental changes in values.

Comparison of future conditions among the archetypes can be informative as they present a continuum of possible future conditions and can highlight the implications to NCP of continuing on the world's current path, or veering to better or worse paths with respect to biodiversity conservation. Consistent with the basic uses of modeling, they can also be used to develop more detailed pathways to different possible futures.

This chapter follows a logical progression, starting from a synthesis of the modeling literature at local scales, to consideration of the empirical evidence of chapters 2, 3 and 4, to consideration of global modeling efforts and their applicability to the Americas. Thus, in section 5.3, literature involving local scales is reviewed and synthesized into the larger context of the regional scale. In section 5.4, we elaborate narratives for the units of analysis based on focal issues of importance to the Americas Region drawn from the information contained in chapters 2, 3 and 4. section 5.5 examines the results of global-level modeling, and how global databases and models can be used in the America's context. Section 5.6 examines present thoughts on particularly important considerations in the development of pathways to a sustainable future. Throughout development of this chapter, we were able to identify clear limits to the modeling approach imposed by lack of data or simply the fact that the modeling has not been done. These "data gaps" provide guidance as to future areas in need of research to generate data for more in-depth and expansive analyses with respect to geography, status and trends of biodiversity and its indicators, and direct and indirect drivers; we consider these, along with our conclusions section 5.7.

A separate IPBES effort is focusing on the concept of sustainable use of biodiversity, and hence, sustainability is not the focus of this assessment specifically. However, when we consider the integration of NCP, trends in biodiversity, drivers, and policy in this chapter, we are doing so with the ultimate purpose of informing not only policy makers, but other IPBES teams with respect to issues related to sustainability. Thus, discussions related to resource exploitation, pollution, and land use change are intimately related to sustainability and will be considered, as appropriate, throughout this chapter.

5.3 Informing the future from local studies

Given the regional diversity of ecosystems, the heterogeneity of social groups, different types of local knowledge and country-based environmental decisions and policies, transformation processes are expected to occur at different magnitudes and in response to the influence of distinct drivers of change throughout the region. Arguably, a precise understanding of future trends of biodiversity and nature's contribution to people for the Americas, the role that different drivers, models and scenarios play in this understanding, and the amount of synergies and trade-offs between them requires the analysis and synthesis of studies developed at local scales.

In an attempt to elucidate what is known concerning the relationships between indirect and direct drivers, direct drivers and nature, and nature and nature's contribution to people, a literature search was conducted to identify studies with a local scope that used a prognostic approach through "modeling" to determine the nature, form and future projections of those relationships. Within this context, models are seen as "qualitative or quantitative descriptions of key components of a system and of the relationships between those components" (IPBES, 2016). We conducted an initial literature review based on Thompson Reuters Web of Science database using an open search approach in which different combination of search terms were used (e.g. scenarios, ecosystem services, biodiversity, participative scenarios, nature's futures, visions, land use change scenarios, climate change scenarios). The search lasted until September 2016. From each document, the abstract was evaluated for its suitability for the chapter where the main criterion was that analyses use projections, trends or narratives into the future. Subsequently other documents were identified through the list of references as well as recommendations by third parties. This led to a selection of 36 local case studies published between 2001 and 2017 (Figure 5.2).

Figure 5 2 **Geographic distribution of the 36 local studies used for the analysis.**

Yellow: social studies; Purple: ecological studies; Blue: economic studies. Source: own representation visualized in google maps.

The consulted literature could be categorized into 3 groups: studies mainly with a social science perspective (accounting for 25% of the total), those with an economic focus (17% of the total) and predominantly ecological studies (58% of the total), aiming at understanding current drivers, indicators and trends in the use of ecosystem services. These groups, however, are not mutually exclusive as some of the studies do apply to more than one category.

The first group, with a predominantly social sciences approach, focused mostly on stakeholders' perceptions and dependence on ecosystem services (Cárcamo et al., 2014; Riensche et al., 2015), community adaptation responses (Brown et al., 2016), the political process in nature conservation (Manuschevich & Beier, 2016), effects of natural phenomena on people and property (Arkema et al., 2013) and social implications of land use change (Evans et al., 2001; Mastrangelo & Laterra, 2015; Tejada et al., 2016).

A commonality in this type of studies is the use of participative approaches for scenario development. In a recent review and analysis of several participative scenario exercises, Oteros-Rozas et al. (2015) grouped different studies according to their application and utility. Studies were placed in each of the four identified clusters as follows:

Cluster 1: studies that performed desirability and vulnerability analysis. These studies broaden the thinking of social actors about social-ecological systems and also identified the stimulation of creative and complex thinking as a strength (Beach & Clark, 2015; Quinlan, 2012; Ruiz-Mallén et al., 2015).

Cluster 2: studies that identified stakeholders and drivers of change before workshops, and developed backcasting during the participatory process. They aimed to understand the social and institutional mechanisms behind management decisions and they recognized insights for landscape management as a positive outcome (Vilardy-Quiroga & González Novoa, 2011).

Cluster 3: studies that identified direct drivers of change prior to participatory scenario planning and explicitly included uncertainty. They aimed to promote community-based solutions and recognized as a positive outcome having engaged social actors that are unrepresented in decision making (e.g. Mistry et al., 2014).

Cluster 4: studies that used modeling as a quantitative technique after a workshop and monitoring processes. They aimed to facilitate sharing experiences among stakeholders in a creative and collaborative way. In this cluster, a complex understanding of the current situation and the co-learning process between scientists and nonacademic stakeholders were highlighted by researchers as positive outcomes (e.g. Peterson et al., 2003; Ravera et al., 2011a, 2011b; Waylen et al., 2015).

The second group, which makes predominant use of economic tools was concerned with the valuation of ecosystem services (Nelson et al., 2009; Outeiro et al., 2014), land use changes (Schneider et al., 2012), combining agricultural productivity with conservation (Latawiec et al., 2014), economically beneficial climate change adaptation strategies (Rosenthal et al., 2013), and forestry and future land use (Radeloff et al., 2011).

The third group's studies discuss issues from an ecological perspective. They encompass issues such as deforestations' causes and effects, landscape fragmentation (Piquer-Rodríguez et al., 2015; Zanella et al., 2012), land use change (Aguiar et al., 2014; Del Toro et al., 2015; Lawler et al., 2014), bioclimatic niches (Giovanelli et al., 2008; Uden et al., 2015; Urbina-Cardona & Castro, 2010; Urbina-Cardona & Flores-Villela, 2010; West et al., 2015), ecological interactions (Bello et al., 2015; Jarnevich et al., 2017), impacts of agriculture on biodiversity (Chaplin-Kramer et al., 2015), effect of anthropogenic occupation to nature and nature's contribution to people (Duggan et al., 2015; van Soesbergen & Mulligan, 2014; Verutes et al., 2014), as well as general effects of agriculture and forestry on nature (Aguiar et al., 2016; Giannini et al., 2015; Müller et al., 2014; Uden et al., 2015). Studies investigating scenarios or future trends of the condition of marine ecosystems are scarce in the Americas but the review analysis of Teh et al. (2016) investigating the future of Canada´s oceans and marine fisheries is a good example to elucidate how environmental change and socioeconomic pathways will play a role on marine ecosystems integrity.

Forty seven percent (47%) of the studies analyzed include a multiple driver approach. The analysis revealed an impressive diversity for both direct and indirect drivers affecting nature. Among them, urbanization, climate change, political process and land use change were the most cited. In general, these local studies show that anthropogenic drivers affect nature and nature's contribution to people both indirectly through policy and directly through immediate changes in nature as caused by such factors as deforestation. Importantly, among the studies, a particularly strong correlation is found for land use change as a driver of deforestation.

Another important finding from the local literature regards to biological invasions that, acting in synergy with climate change, are predicted to increase areas suitable for exotic species such as reptiles like *Lithobathes catesbeianus* (Bullfrog) in Brazil and Colombia (Giovanelli et al., 2008; Roura-Pascual & Suarez, 2008; Urbina-Cardona & Castro, 2010). By 2050, *Hemidactylus brookii* (now *H. angulatus*) and *Hemidactylus turcicus* could increase their range by 72.6% and 33.5% of Colombia's area, respectively.

The most common indicator to measure human's impacts on nature across the analyzed studies was deforestation, second was biodiversity loss. Although, the diversity of indicators was large among the analyzed studies.

With regards to indicators, the first group of studies used indicators of nature´s contribution to people such as freshwater quality, climate regulation, aesthetic values, value of biodiversity and resource availability. The value of ecosystem services and productivity were also found as indicators. Human well-being indicators were human vulnerability to natural disasters and dependency on ecosystem services.

The second group of studies used mostly monetary valuation of ecosystem services as an indicator. Typical economic indicators were land use and economic benefits of land use change as for example the shifts from agricultural to urban land use and cover (Schneider et al., 2012).
The third group of studies mostly presented ecological indicators such as change in forest cover and connectivity, deforestation dynamics, species distribution, biodiversity, carbon storage and emissions, change in species compositions and abundance, and effects of anthropogenic activity on nature, such as water quality.

Among the studies, the most common trends linked to the "economy prevails" archetype were biodiversity loss due to agriculture or forestry and the negative impacts of urbanization. The positive impacts of more strict environmental conservation legislation found in the studies can be linked to the "policy reform and great transition" archetypes.

Studies showed very clear negative effects on nature by urbanization, intensified agriculture (Chaplin-Kramer et al., 2015; Müller et al., 2014) and forestry, energy production and climate change. However, by changing to sustainable agricultural practices, productivity could be increased with less impact to biodiversity (Latawiec et al., 2014). One important recommendation found is that in political processes, the relationship between political dynamics and economic processes, communication and early stakeholder engagement as well as more equitable access to ecosystem services should be addressed by decision makers (Cárcamo et al., 2014; Manuschevich & Beier, 2016).

In summary, the biggest challenge informing regional futures of nature and nature's contribution to people from local studies is that the limited number of studies, different methodologies and heterogeneity (in terms of indicators, drivers and trends) produce a number of different results. This makes scalability (from local to regional) a challenge yet to overcome. There is a clear need for the production of comparable studies at the local level that can aid to better understand the region. Narratives scenarios at the local scale, similar to the ones developed by the Global Environmental Outlook-6 for Latin America and the Caribbean, could well bridge this gap. Despite current scarcity of such studies, it was possible to draw preliminary findings on how the region can be informed through local studies. For example, the presence of agriculture expansion leading to loss in biodiversity illustrates a common trend from various local studies suggesting plausible scalability.

In conclusion, there are two major issues that emerge:

(1) Although models can be a powerful tool to integrate and synthesize the complex dynamics of coupled human and nature systems, a major gap on modeling and scenarios, identified from the literature review is

related to the lack of studies integrating changes in nature with changes in NCP and good quality of life. Consequently, the complexity of these interactions and feedbacks are still not fully represented in the models.

(2) The second issue to point out regarding the current understanding of the relation between human and nature through modeling and future scenarios, concerns the scale and feedbacks considered in the analysis. Global models represent quite well broad trends and analysis, however, there remains a gap in downscaling this information and the feedback from the global approach to the regional and local: a gap to be filled in the future. As well, local studies, representing specific trends in a specific unit of analysis is not frequently upscaled to larger areas. Within this same logic, issues of telecoupling are not well represented either.

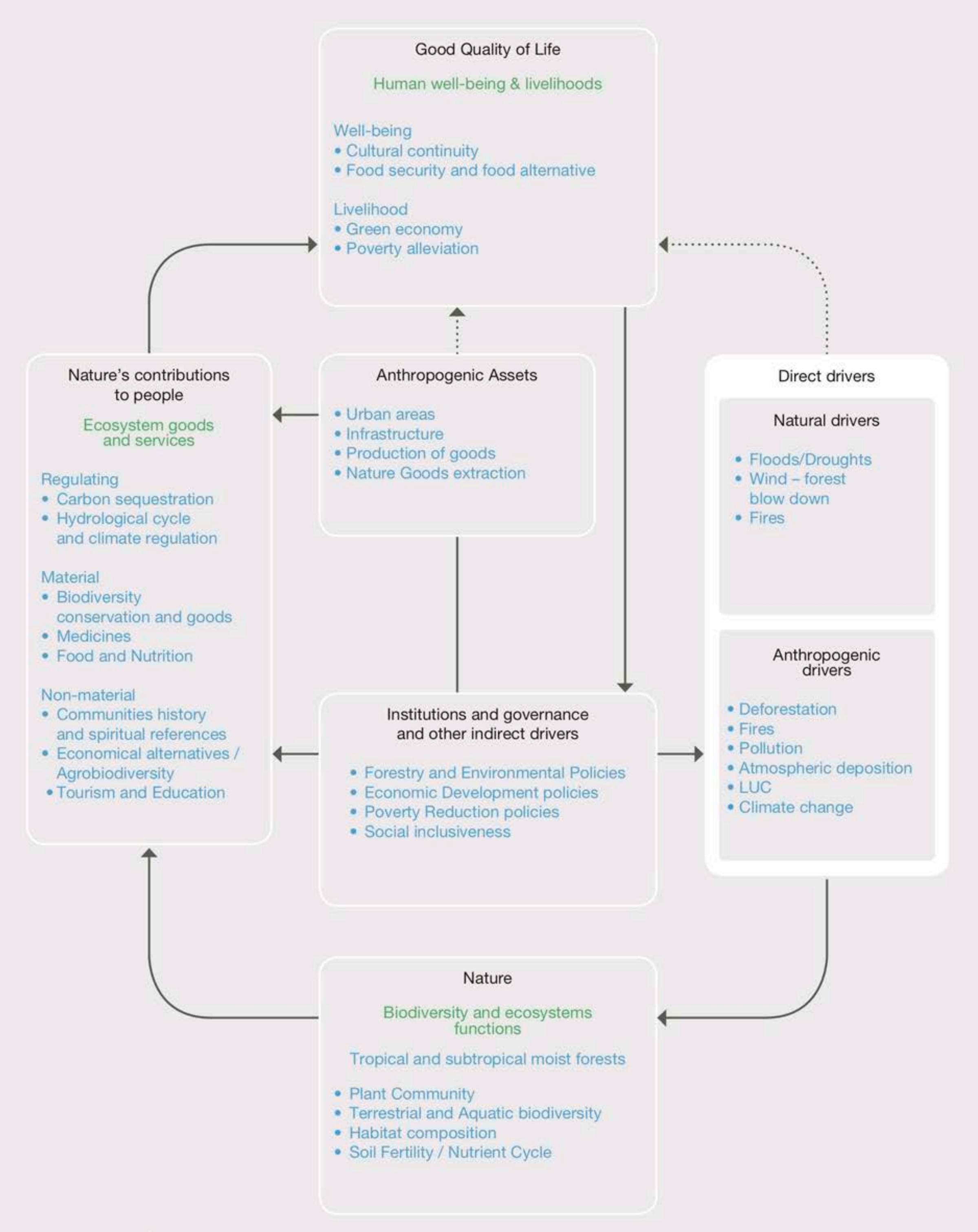
5.4 Informing the future from regional studies: Focal issues within units of analysis and other ecological systems

This section presents syntheses of the information contained in Chapters 2, 3, and 4; focusing on key issues within the IPBES framework. As it is not possible to comprehensively consider all of the units of analysis within each subregion, and that the units of analysis do not address some commonly recognized socioecological systems important in the Americas, we present the information at the regional level, and in the narratives, we concentrate on specific issues that we feel are illustrative of the issues in general. With respect to the information contained in the figures based on the IPBES framework, for NCP, indirect drivers, and direct drivers, the primary bullet items follow the nomenclature and taxonomy of the issues as presented in Chapters 2, 3, and 4. However, for the sub-bullets, as well as the boxes corresponding to quality of life, anthropogenic assets, and nature, we used the terminology as cited or interpreted from the literature. While this results in a profusion of terms, it also gives a sense of the lack of consistency in describing drivers and NCP in the literature; we felt this appropriate in order to convey the many ways that these factors are viewed and referred to.

5.4.1 Tropical and subtropical dry and moist forests UAs – Trade-offs between multiple ecosystem goods and services and scale effects

5.4.1.1 Tropical and subtropcial moist forests

Figure 5 3 Tropical and subtropical moist unit fof analysis viewed in the IPBES conceptual framework. Source: own representation.



Forests are extremely important ecosystems because of their multiple functions in biodiversity conservation and ensuring long-term environmental stability, while providing a variety of economically-important products and services (De Costa, 2011). Tropical forests cover 10 % of all land area (i.e. 1.8×107 km^2) (Mayaux et al., 2005), and represent about half of global species richness. Clearing of these forests is estimated to account for 12 per cent of anthropogenic carbon emissions (Dirzo & Raven, 2003). Over half of the tropical-forest area (1.1×107 km^2) is represented by moist tropical forests (also called 'moist tropical forests', 'wet tropical forests', or 'tropical rainforests'), characterized by high tree-species diversity and high biomass density (Ter Steege et al., 2003).

Asner et al. (2009), alarmingly wrote "In recent decades the rate and geographic extent of land-use and land-cover change has increased throughout the world's moist tropical forests. The pan-tropical geography of forest change is a challenge to assess- and improved estimates of the human footprint in the tropics are critical to understanding potential changes in biodiversity. We combined recently published and new satellite observations, along with images from Google Earth and a literature review, to estimate the global extent of deforestation, selective logging, and regrowth in moist tropical forests. Roughly1.4% of the biome was deforested between 2000 and 2005". According to the Global Forest Resources Assessment (FAO, 2015) of the Food and Agriculture Organization of the United Nations (FAO), compiled by Keenan et al. (2015), indicate that, in the period from 1990 to 2015 Central America lost 25% of forest cover, South America lost 10%, North America gained 0.4% and the Caribbean gained 43%. At global level, the tropical forest suffers the biggest pressure, with higher deforestation rates. Despite the reduction in the past 25years, deforestation is still in high levels. In the period from 1995 to 2000, the raters were at 9.54 million/hectares/year, while from 2010-2015, the rates fell to 5.52 million/ha/year (Keenan et al., 2015). Carbon emissions from tropical deforestation were at the range of 2.9 $\pm$ 0.47 PgC/year during the period from 1990-2007 (Pan et al., 2011). From the period from 2000 to 2005, Asner et al. (2009), estimated that about 20% of the moist tropical forest biome was undergoing some level of timber harvesting, and that forest regeneration on this unit of analysis was basically occurring in hilly, upland, and mountainous environments, which are areas considered marginal for large-scale agriculture and ranching. Aside from deforestation, another growing threat to moist tropical forests, especially to indigenous land and protected area, is mining (Ferreira et al., 2014; Boillat et al., 2017).

For biodiversity however, droughts, coupled with increased evapotranspiration from rising temperatures, can cause forest dieback expressed as the loss of both carbon and tropical species (Oliver L Phillips et al., 2009). Moreover, there is a significant likelihood of future forest dieback in the Amazon under most climate change projections (Malhi et al., 2009). The future of moist tropical forests has become one of the iconic issues in climate-change science (Zelazowski et al., 2011). For instance, the extensive tropical rainforests of Amazonia affect the functioning of the Earth's climate through the exchange of large amounts of water, energy, and carbon with the atmosphere. During the past few decades, a large research effort has been devoted to understand the functioning of Amazonian ecosystems and their responses to deforestation, climate change, and altered fire regimes (Gloor et al., 2015). Changes in forest species composition, increasing dominance of lianas and turnover rates have been reported (Laurance et al., 2004; Lewis et al., 2004; Phillips et al., 2004). Based on an extensive field site network, Brienen et al. (2015) suggest a strong decrease in the Amazon forest net carbon sink. The increase on the frequency of extreme drought events was suggested to worsen these responses in the future (Feldpausch et al., 2016). Moreover, there is a significant likelihood of future forest die back in the Amazon under most climate change projections and it is uncertain which species will adapt to novel climates projected to concentrate in tropical forest biomes (Zemp et al., 2017). The main negative effects of the increasing climate variability on forests will likely be via occasional drier and hotter episodes particularly in those regions which have experienced a slight drying trend, i.e., the southwest and south of the basin (Gloor et al., 2015). Seasonality and strength of carbon fluxes in the Amazon forest might be affected, in the short term, by climate change (Gatti et al., 2014).
Aside from the fact that deforestation and forest degradation is the biggest threat for forest areas in the tropics, (Bustamante et al., 2016), some studies show a tendency of the potential extent of moist tropical forests in future climate regimes between 2°C and 4°C, where a risk of forest retreat, especially in eastern Amazonia and Central America are highlighted. The main conclusion is that the water availability is the best determinant of the current distribution of moist tropical forests, which can dominate over other vegetation types only in high-precipitation, low water-stress environments; the change in the extent of the moist tropical forests niche is uncertain (Zelazowski et al., 2011). Some global circulation models predict increase

in drought frequency in the South American Amazon (Cox et al., 2004); however few experimental data simulate the Amazon response to climate change (Davidson et al., 2012). With lack of experimental data and the complexity of the forest ecophysiological process, in response to change in temperature and precipitation (mainly parameters simulated by global circulation models), models a decade ago simulated a dramatic amazon forest die back (Cox et al., 2004). More recently a strong resilience of the Amazon forest has been suggested by simulations, much associated with the positive vegetation primary productivity response to the increase in the atmospheric carbon dioxide (Cox et al., 2013; Huntingford et al., 2013). Anadón et al. (2014) found that climate change will increase savannas at the expense of forests and treeless vegetation in tropical and subtropical Americas (Figure 5.4), predicting a large shift in the savannah-forest transition in the eastern Amazon, supporting the hypothesis that climate change will lead to more unsustainable states for theses ecosystems (Figure 5.5).

However, the key message remains related to the ability of moist tropical forests to acclimate and adapt to future temperature changes. De Costa (2011) suggested that due to the narrower range of seasonal temperatures experienced by forests in the moist tropics, the capacity to adapt is considered to be lower than that of temperate forests. Indicative of this pattern is the reduction in sequestration of carbon observed during years of warmer temperatures and lower precipitation resulting from El Niño Southern Oscillation (De Costa, 2011) or even stronger seasonal patterns (Gatti et al., 2014).

Figure 5 4 **Transition map for the forest–savanna system for the present time (1950–2000) and for the year 2070 under the RCP8.5 scenario in the tropical and subtropical Americas. Source: Anadón *et al.* (2014).**

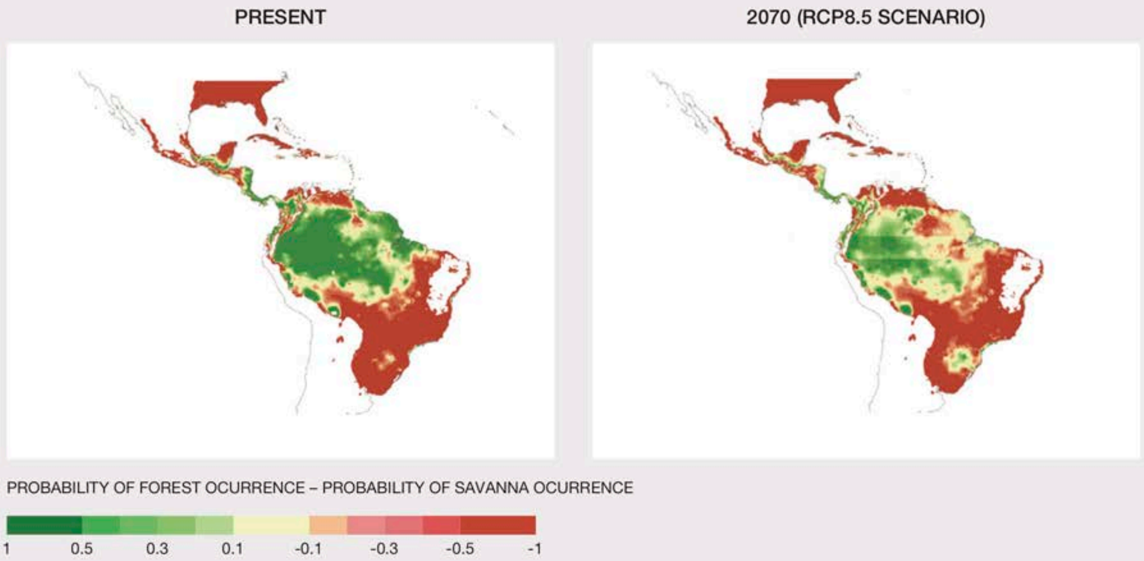
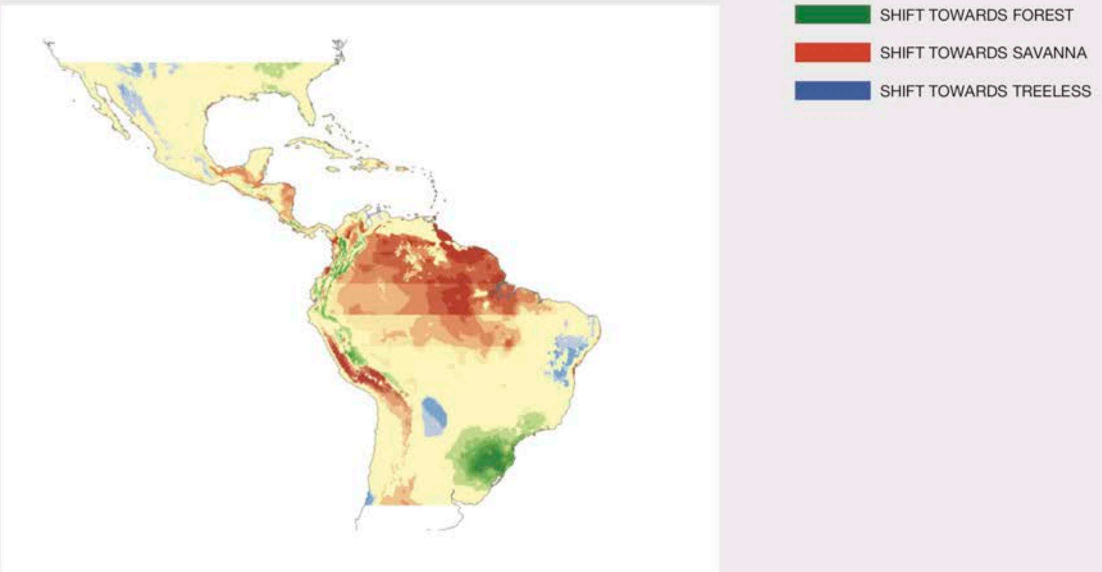
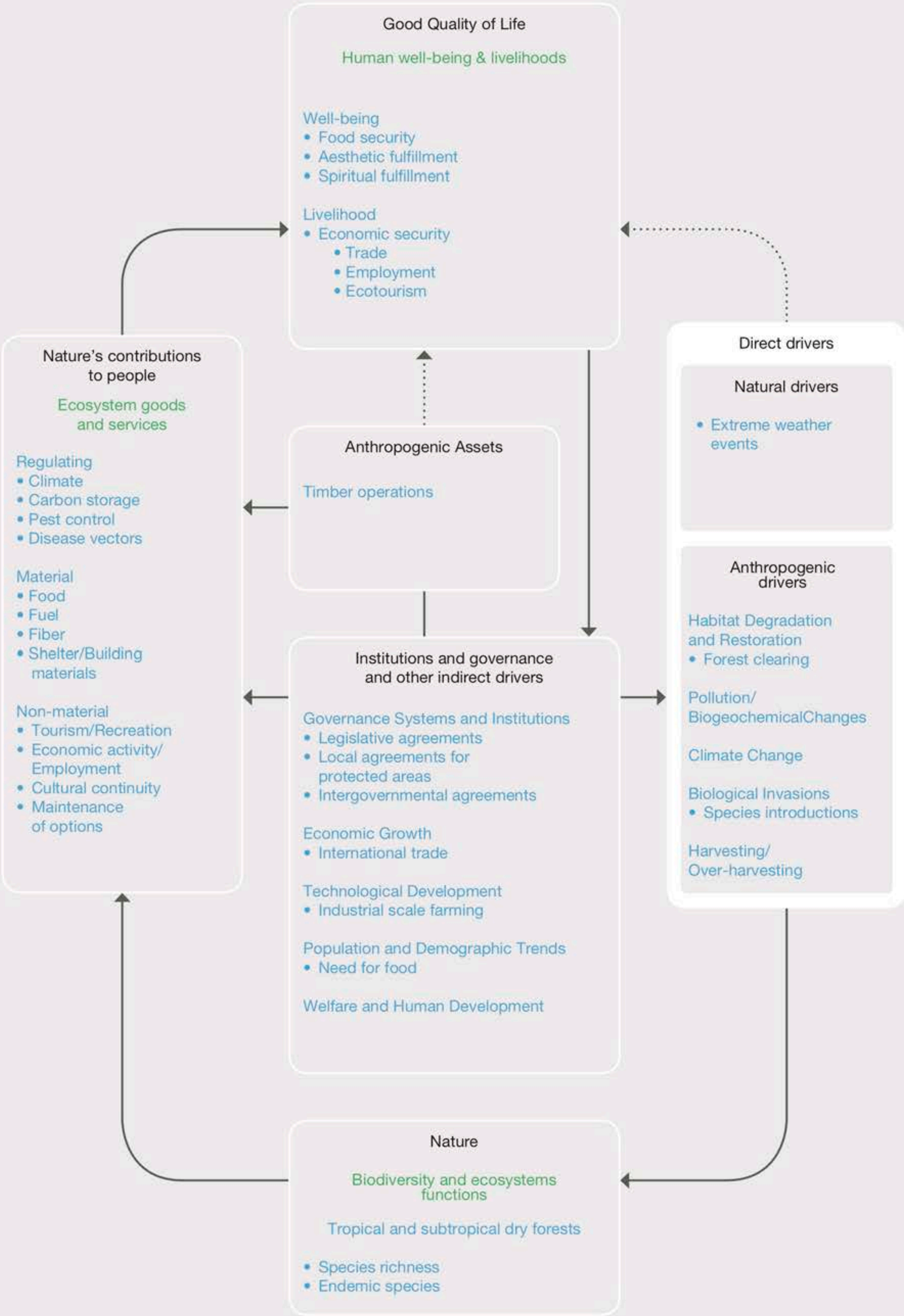


Figure 5 5 **Projected shift towards forest, savanna or treeless states for the year 2070 under the RCP8.5 scenario in the tropical and subtropical Americas. Source: Anadón *et al.* (2014).**



5.4.1.2 Tropical and subtropical dry forests unit of analysis

Figure 5 6 Tropical and subtropical dry forests units of analysis viewed in the IPBES conceptual framework. Source: own representation.



Tropical dry forests occur from Mexico, through Latin America and the Caribbean, with the most extensive area being the Gran Chaco of South America. The forests contribute to human well-being on a local scale through regulating services, such as erosion control and micro-climate regulation, and provisioning services, such as non-timber forest products (e.g. bushmeat, fodder, and firewood), and non-material NCP such as cultural identity. However, these services are becoming increasingly impacted due to land conversion that replaces these locally-relevant services by services relevant on larger scales, e.g commodity agriculture (Lapola et al., 2013). Thus, changing global demographics, consumption patterns, and global trade are driving land conversion from tropical dry forest to other uses such as cropping and cattle ranching, leading to the loss and fragmentation of native ecosystems. These land-use changes produce a strong trade-off between ecosystem goods such as grains and beef for export and the regional or country level, economic benefit, versus ecosystem services relevant for local people. Further discussions are found in Chapter 3.

The evolving trade-off underscores one of the main challenges inherent in sustainable use of biodiversity, namely spatial scales as relevant to the generation of ecosystem goods and services as opposed to where their benefits are ultimately realized. These scale considerations include local social-ecological systems where ecosystems are converted and local population is displaced, the national scale where the different Chaco countries (Argentina, Bolivia, Paraguay), or Cerrado (Brazil), design and implement their agricultural and environmental policies, the regional scale where some environmental processes become relevant (e.g. climate regulation) and the global scale where driving forces originate (China´s demand for soybean meal to feed pigs and poultry) and where countervailing policies may be created (e.g. Reducing Emissions from Deforestation and Forest Degradation).

Thus, effective policies for addressing the conversion of dry tropical forest to other uses will need to be addressed at various organizational scales. National governments affect land-use changes through agricultural (e.g. technology adoption), economic (e.g. currency devaluation, reduction of fiscal pressure) and environmental policies (e.g. land-use planning); companies and corporations that operate along the agro-industrial chain influence the rate and direction of land-use changes; international organizations (e.g. Roundtable for Responsible Soy) lobby national governments to increase or decrease agricultural expansion over native forests, etc. The policy and environmental challenges are to define effective and sustainable land use planning, which includes strong institutional arrangements, clear legislation and economic opportunities for conservation and sustainable production.

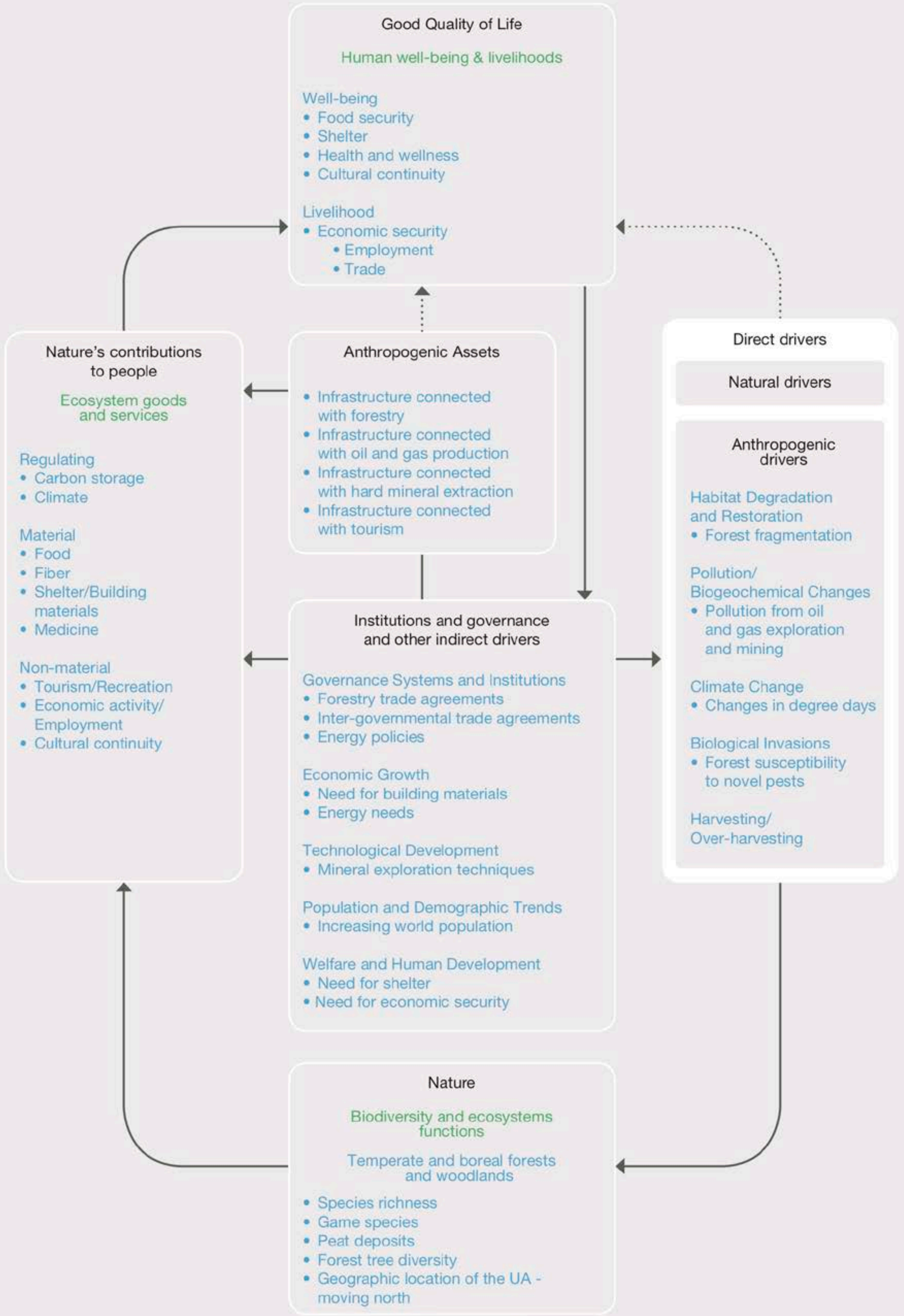
Just as there is a significant component of temporally changing demographics and consumption patterns, there are other temporal aspects to this issue, including the temporal considerations inherent to this unit of analysis. The decadal scale is relevant for climatic fluctuations (e.g. dry and wet periods) that naturally occur in the Gran Chaco and that strongly affect agricultural production. At the scale of centuries there may occur fluctuations in ecosystem state, such as changes in the dominant vegetation, with periods of woodland domination being followed by periods dominated by herbaceous (savanna-like) vegetation. Within periods dominated by woodlands like the current one, regeneration of dominant tree species (e.g. *Prosopis* spp., *Schinopsis* spp.) after land conversion may take more than 50 years due to the slow growth rate of these species.

The United Nations Environment Programme (UNEP, 2016a) considers three scenarios for Latin America and the Caribbean: ‘economy prevails’ scenario tends to maximize economic growth at the expense of social and environmental objectives. This approach is reactive in terms of policy responses. Consequently, economic growth instability increases, as does vulnerability to unforeseen events. Policy options in this outlook emphasize privatization of public services and attempts to internalize environmental and social externalities into the costs of production through market tools. On a ‘Policy trade-offs’ scenario, new policies and regulations are introduced to partially mitigate the adverse impacts of more than two decades of neo-liberal practices, in this scenario, population growth slows, urbanization stabilizes and emigration pressures reduce. The policy trade-offs scenario promotes greater transparency, policy effectiveness, and institutional coordination. However, environmental sustainability, even while a policy objective, remains a secondary priority for governments. Finally, a ‘sustainability agenda’ scenario assumes the implementation of policies to promote sustainable approaches to agricultural practices, rather than market signals, more conscientious tourism, and a more participative and coordinated strategy for energy trade. However, in some areas, this outlook may result in a slowing of technological intensity, as well as a shift towards local-level issues. In this case, policy options tend to prioritize the emphasis on building and keeping a social consensus through education and institutional strength (UNEP, 2016a). Whether considering spatial or temporal scales, the inherent trade-offs or synergies associated with this issue need to be considered fully.

These trade-offs include: forest loss and fragmentation increases agricultural area and production volumes at the expense of biodiversity; forest degradation increases accessibility of cattle to natural fodder, but decreases carbon sequestration on biomass; landscape homogenization facilitates agricultural operations but reduces livelihood options for local people, forcing them to migrate into urban areas, etc. Regardless of the ultimate trade-offs, this issue is urgent in that tipping points may be reached that eliminate a reasoned approach to the trade-offs, such as: regarding climate, the loss of forest cover alters the hydrological cycle and forces the system towards drier conditions; regarding vegetation, the degradation of woodland vegetation alters soil and climate conditions and shifts the system towards one dominated by scrublands.

5.4.2 Temperate and boreal forests and woodlands units of analysis – Key to indigenous people and carbon storage

Figure 5 7 Temperate and boreal forests and woodlands units of analysis viewed in the IPBES conceptual framework. Source: own representation.



Temperate and boreal forests occur in the northern hemisphere of the Americas – mostly in the USA and Canada. The boreal forest covers northern Canada and Alaska with a belt of coniferous forests. Boreal forests, and the peatlands that many grow on, are critical for carbon storage. Temperate forests are located in eastern North America. They are comprised of a mix of deciduous, broadleaved and coniferous evergreen forests. Temperate rainforests – which are dominated by coniferous trees - are found on the on the West Coast of North America in British Columbia and in the USA´s Pacific Northwest. In addition, evergreen rainforest occurs in Chile.

Boreal forests are known for caribou (*Rangifer tarandus groenlandicus*), moose (*Alces alces*), bear (*Ursus* spp.), beaver (*Castor canadensis*), rabbit and migratory birds, which are important to local and Indigenous communities. Indigenous communities have lived in the boreal forest for thousands of years. There are more than 600 primarily indigenous communities in the Canadian boreal region. They rely on the forest for physical subsistence and cultural wellbeing. Fish and waterfowl provide for a significant part of the subsistence diet for many remote communities.

In addition to the cultural and provisioning benefits provided to local populations (Figure 5.7), carbon storage is a key NCP. Climate change, which is considered the primary anthropogenic driver in this system, has resulted in temperatures changing faster in the high latitudes than in any other area on the planet (IPCC, 2013a).

The boreal landscape is dominated by an active natural disturbance driven by large area stand-replacing wildfire and insect outbreaks (Price et al., 2013). Changes in climate, atmospheric carbon dioxide concentrations and fire regimes have been occurring for decades in the global boreal forest. Future climate change is likely to increase fire frequency and insect outbreaks. Warming in the boreal region is projected to be substantially above the global average. According to the IPCC (Intergovernmental Panel on Climate Change), temperatures in the northern boreal have increased at twice the global rate. Boreal forests are particularly sensitive to warming because of their soils (e.g. peat, permafrost) and likelihood of increased incidence of fire disturbance.

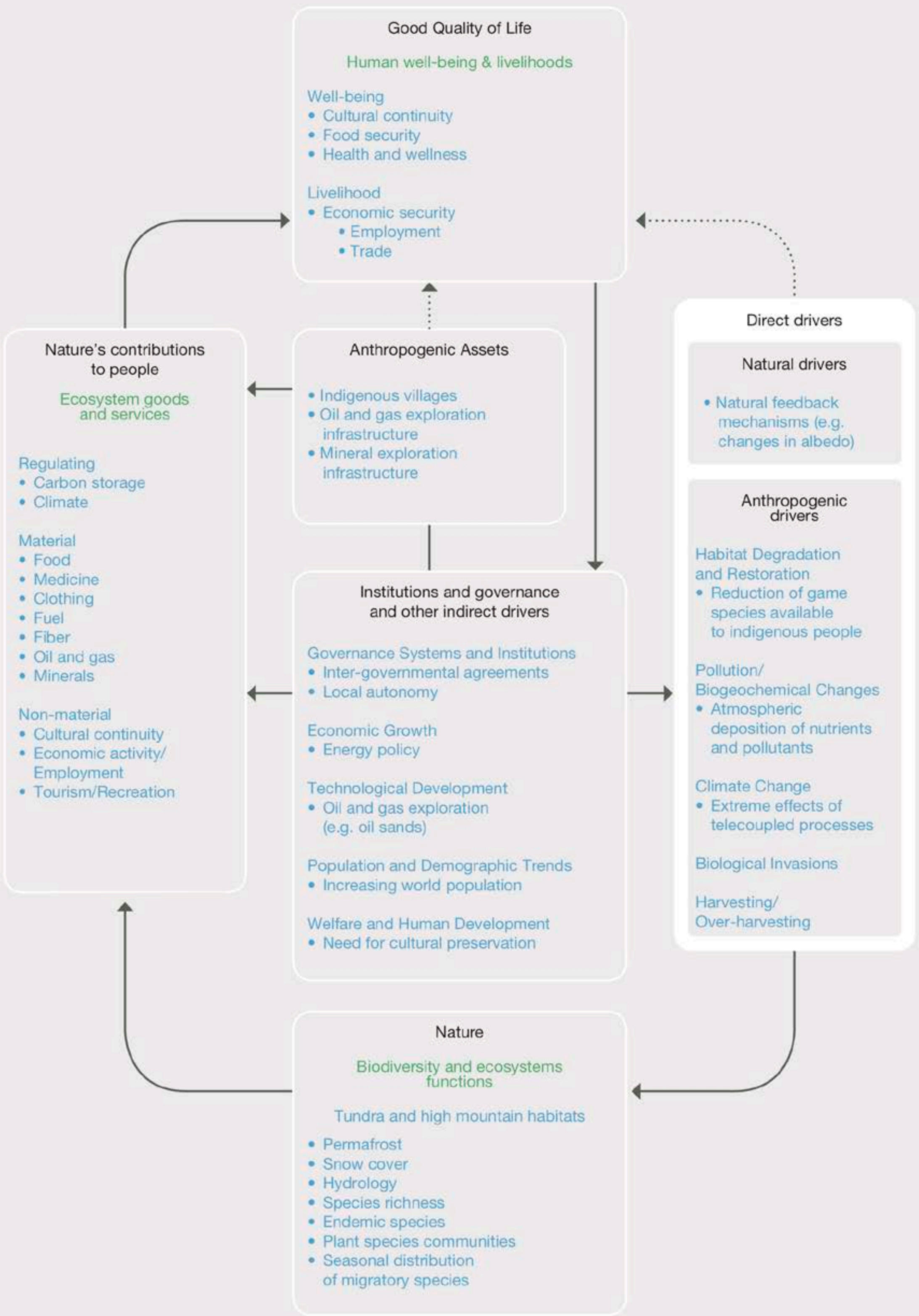
Predictions of future climate largely agree that Canada’s boreal forests will experience substantial warming (Plummer et al., 2006). Lenton (2012), argues that the boreal forest (and arctic) is subject to a tipping point due to strong internal feedback systems; an increase of 4°C global warming (7°C above current levels in the forest) will result in a marked increase in susceptibility to disease. If such a tipping point is reached, there could be significant changes in the landscape (i.e. tree die-off, conversion to grassland) and release of carbon.

Resource extraction, oil and gas development, and timber harvesting are increasingly fragmenting the boreal region, which is impacting migratory connectivity, ecosystem integrity, habitat resilience and species diversity, especially for migratory species. Additionally, the role of infectious plant diseases, mediated by invasive species, will continue to be a significant issue negatively affecting the temperate forests in the future (Chapter 3).

Boreal forests are experiencing the most rapidly changing climate (along with tundra) anywhere on Earth and are likely to be impacted in critical ways in coming decades. Predicted climate change is anticipated to cause shifts in species ranges, with an average northward shift of about 700 km for Canadian tree species; with some species expected to shift as much as 1000 km northwards (sugar maple (*Acer saccharum*), black willow (*Salix nigra*), American basswood (*Tilia americana*) and white alder (*Alnus rhombifolia*)) (McKenney et al., 2007). Biodiversity gains are anticipated in Canada's maritime provinces, including Quebec, Ontario, northern prairies and Alaska, with up to 60 new tree species possibly appearing in some areas, although low soil fertility might limit their migration (McKenney et al., 2007). Extreme fires in intensity and extent have threatened forests in recent decades partially as the result of forest management practices that have permitted decades of deadwood (fuels) to accumulate (Oswalt & Smith, 2014). Drought is exacerbating wildfires in western forests, particularly in California in the USA and Alberta in Canada.

5.4.3 Tundra and high mountain habitats units of analysis – Remote, but not remote enough

Figure 5 8 Tundra and high mountain habitats units of analysis viewed in the IPBES conceptual framework. Source: own representation.



Tundra occurs in two settings within the Americas; at high elevations ("alpine tundra") and in the high latitudes ("arctic tundra"). Arctic tundra is circumpolar in its distribution and accounts for a large amount of land area across the USA and Canada (Chapter 2). Adjacent marine areas of the arctic are also critical habitat for numerous tundra species. It presents a unique set of circumstances with respect biodiversity and NCP, namely that of all the units of analysis, Tundra is the most closely linked with respect to NCP and local ecosystems and that the primary drivers affecting the system are almost wholly external to the region in which the unit of analysis occurs. Tundra includes well-known fauna, such as barren ground caribou and muskoxen (*Ovibos moschatus*), which are important to indigenous populations from subsistence and cultural standpoints (Figure 5.8). While difficult to separate, it is perhaps this latter consideration that is the primary NCP for Tundra, for while the physical needs of the indigenous people associated with Tundra could, conceivably, be replaced with market goods, the culture of these peoples is intimately related to biodiversity of the system; loss of which would threaten the cultures continuity.
Aside from the cultural and provisioning NCP accrued to local populations depicted in Figure 5.8, the NCP of carbon storage is of concern on a global basis. Arctic tundra is estimated to store approximately 50% of the world's soil carbon (Tarnocai et al., 2009), mainly in the form of permafrost (perpetually frozen soil). But climate change, which is considered the primary driver in this system, has resulted in temperatures changing faster in the high latitudes than in any other area on the planet (IPCC, 2013a) (Chapter 4). Thus, the situation with respect to Tundra provides a clear example of telecoupling, i.e. where cause and effect are separated geographically, but are clearly related.

The warming in the Tundra, and its neighbouring marine areas, has resulted in several changes that have affected the Tundra, including: thawing of the permafrost (Walker et al., 2006), changes to the plant communities (reduction of graminoid species in favour of shrubs and expansion of the boreal forest) (Hu et al., 2010; Lloyd et al., 2003) (Chapter 3), increased frequency of fires, and changes in neighbouring sea ice conditions (Bhatt et al., 2010). These changes result in a lowering of the local albedo (the reflectivity of the Earth's surface) in the immediate area in the case of shrubs and fires resulting in a positive feedback to climate change and thawing of the permafrost. Due to the amount of carbon stored in the permafrost, the change of tundra from a carbon sink to a carbon source is also of great concern from a global perspective. Adding to the concern is the consideration that warmer temperatures and change in vegetation adds uncertainty to what was considered a relatively stable biome. This uncertainty stems from unknowns regarding the natural processes associated with the tundra. For example, fires which were once rare in the tundra may be increasing in frequency and perhaps extent (Hu et al., 2010) and these fires may increase the rate of stored carbon release (Mack et al., 2011). Additionally, as with the Boreal Forest, Tundra is also subject to a tipping point or threshold with loss of the native plant communities whenever 1000 degree days is exceeded (IPCC, 2014; Lenton, 2012).

Uncertainty is also associated with respect to existence of a "tipping point" with respect to degradation of the permafrost, i.e. a point at which the degradation is irreversible and accelerates (IPCC, 2014). Some modellers believe that such a tipping point exists and that it could be reached within the next 100 years (Scheffer et al., 2012). If such a tipping point is reached, there would be a massive release of greenhouse gases. In that event, it is anticipated that over time the area currently occupied by arctic tundra would be replaced by boreal forest. The implications of this scenario are that the rate of climate change would increase, flora and fauna would be further endangered or driven to extinction and the cultures and traditional ways of indigenous people throughout the Holarctic would be severely impacted.

The issue of melting permafrost and its implications is a particularly intransigent problem for several reasons. The ultimate source of the drivers affecting the system are not internal to the system, rather, they originate faraway geographically, i.e. anthropogenic greenhouse gasses, and are exacerbated through the effects of a positive feedback acting locally and through teleconnection.

With temperatures in the Arctic rising twice as fast as the global average, climate threatens to alter biodiversity and ecosystem functioning in Tundra in the coming decades (Pithan & Mauritsen, 2014; Screen & Simmonds, 2010). Vegetation models predict significant northward range expansion of boreal species into Tundra, leaving few refugia for tundra-specialist species by 2050 (Kaplan & New, 2006; Pearson et al., 2013; Hope et al., 2015). Thus, while the intrusion of boreal species may augment species richness in Tundra, the potential extinction of tundra-adapted taxa may detract from it (CAFF, 2013; Chapin et al., 2000). The overall balance of these processes is uncertain. As sea ice declines, shipping in the Arctic may be a dispersal mechanism for invasive species (CAFF, 2013). Many future changes in Tundra are predicted to be rapid nonlinear transitions, rather than smooth gradual changes. Among such “regime shifts,” the Arctic Council (2016) predicts decreased carbon storage capacity, drying soils, and increased woody vegetation. Experimental and modeling work from several authors across Arctic Resilience Assessment document (Artic Council, 2016) support for these conclusions (for carbon storage, see Abbott et al., 2016; Hu et al., 2015; Lara et al., 2017; Li et al., 2014; Mack et al., 2004; Natali et al., 2015; Schuur et al., 2013; Schuur et al., 2015; Sistla et al., 2013; Sitch et al., 2007; Sweet et al., 2015; Webb et al., 2016).

Treeline advance in North America will continue to reduce the extent of alpine habitat (Harsch et al., 2009), while deciduous shrub growth and overall plant productivity above treeline will increase due to warming (Raynolds et al., 2014). Habitat degradation may also occur through nitrogen deposition (Dentener et al., 2006), with the potential to reduce species richness (Walker et al., 2006).
Distribution modeling predicts northern Andean birds will lose 30-40% of their ranges with compositional changes (Velasquez-Tibata et al., 2013); páramo and puna are predicted to experience reduced species richness and species turnover (Ramirez-Villegas et al., 2014). The biodiversity of hyper-arid alpine areas, where many species depend on moisture supplied by peat bogs could be especially vulnerable. A recent assessment for páramo (Buytaert et al., 2011) concluded that changes in precipitation patterns, increased evapotranspiration and alterations of soil properties will have a major impact on water supply, which will further affect species composition. Warming is expected to have a major impact on seasonal water flow all along the Andes due to loss of glaciers, although the latter will depend on future precipitation trends along the Andes (Vuille, 2013). However, given the complex landscape and regional climatic variation, there are large uncertainties regarding the responses of high Andean biodiversity and ecosystem functions to climate change.

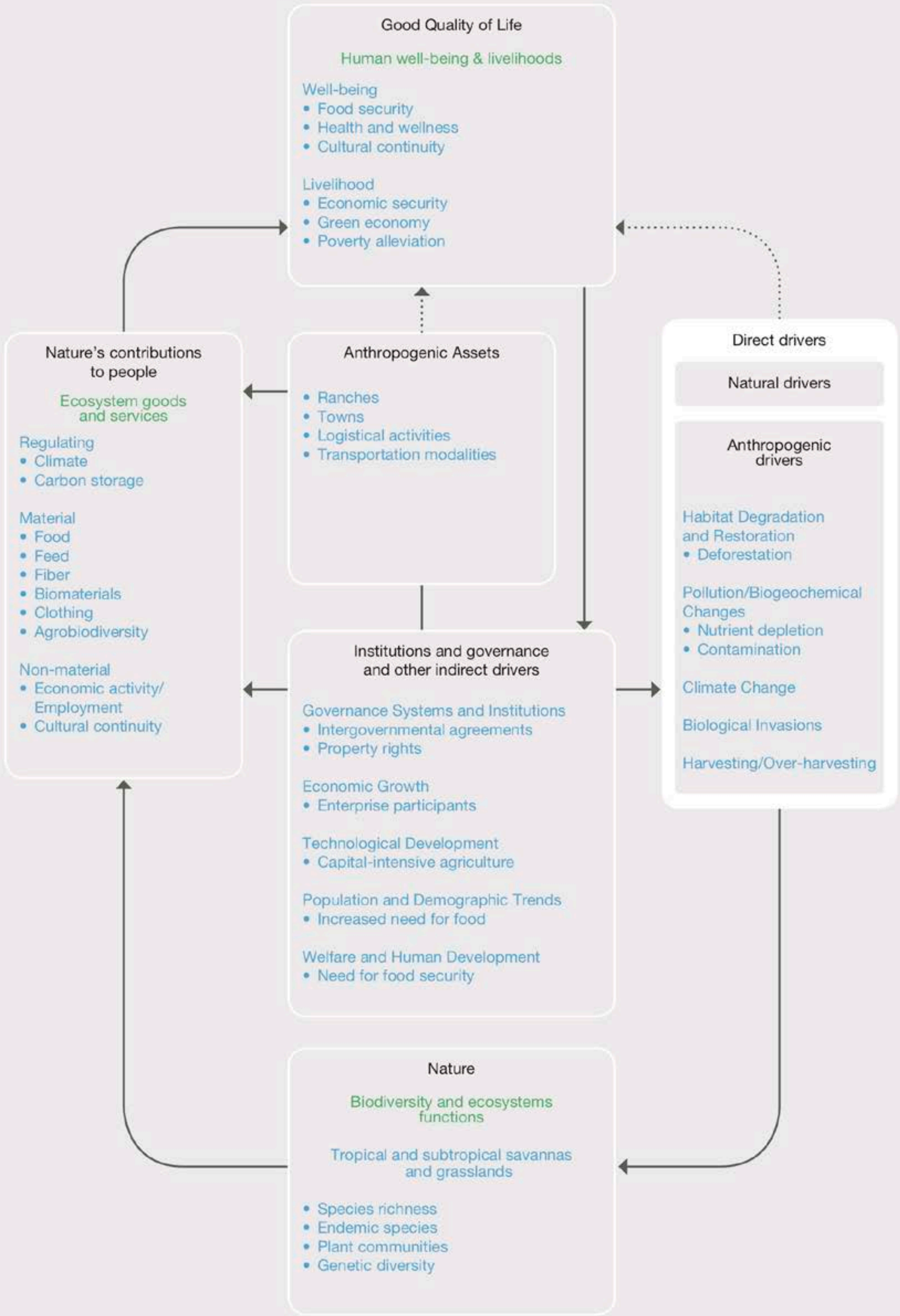
The possible futures for the tundra under the scenario archetypes is somewhat limited due to the facts that the indirect and direct drivers at play are remote relative to the region and the fact that climate change effects in terms of temperature change are more extreme for this region than any other on the globe. Under the Market Forces archetype we can expect the continued reduction of sea ice and thawing of the permafrost to continue as this simply represents a continuation of the factors that have resulted in the impacts seen thus far. Under the Fortress World, archetype we can expect to see a more rapid deterioration of the permafrost and perhaps surpassing of a tipping point with respect to greenhouse gasses release due the ecological processes inherent to the Tundra. The Policy Reform archetype scenarios could be a significant contributor to lessening of the factors at play in the Tundra, but given the fact that climate change effects appear to be greatest at the high latitudes, a very concerted effort would have to be made to adopt policies lessening or reversing greenhouse gasses emissions. Because of the telecoupling and teleconnection aspects involved with the tundra, this scenario would require a coordinated effort on a global scale, as there is little that local populations and policymakers can do to affect the drivers involved. This latter consideration, namely that an effort on a global scale is needed, argues that to truly avoid a tipping point in the Tundra, an approach within the Great Transition archetype will be required.
Northern ecosystems are highly dynamic and variable, however, climate change is considered to be increasing the nature and range of variability and adding new kinds of stresses that are outside what is considered ‘normal’ as defined by both scientists and indigenous and local knowledge (ILK) (Huntington et al., 2007). This is likely to continue with implications for arctic biodiversity and Indigenous communities that depend on Tundra for their culture and livelihoods. While Indigenous communities are highly adaptive, options for tundra as a biome are limited. In other regions and for other units of analysis, natural adaptation by the biome is possible… arid areas may expand, temperate forests may move north, animals may shift their range along with changing climate envelopes, as have small mammals in North America (Myers et al., 2009). However, as tundra is already at the extreme reaches of the globe, such adaptive responses are limited to non-existent.

Box 5.1. Dealing with Ecological Variability and Change in Human-Caribou Systems

Indigenous communities from tundra (arctic and sub-arctic) regions of Canada and the USA are highly dependent on barren ground caribou (*Rangifer tarandus groenlandicus*) as a foundation of culture and livelihood. There are between 10-15 subpopulations of barren ground caribou in northern Canada and Alaska; both science and ILK tell us these populations tend to rise and fall in a 40-70 year cycle. Although there is much adaptive capacity within northern communities based on ILK, climate change as well as resource development are creating new stresses on human-caribou systems. For example, the Bathurst caribou, which last peaked at 475,000 animals, has declined by 90%, which has had dramatic implications for the diets and well-being of local Inuit, Dene and Metis peoples. Booms in mineral resource development such as diamond and rare earth metal mining, in the absence of a cumulative effects framework will lead to major challenges to arctic biodiversity as well as the sustainability of arctic peoples and livelihoods. The preservation of these resources for use by indigenous people is a major goal in this region (Environment Canada, 2016; Gunn et al., 2011; Parlee et al., 2013).

5.4.4 Tropical and subtropical savannas and grasslands unit of analysis – Agriculturalization

Figure 5 9 Tropical and subtropical savannas and grasslands unit of analysis viewed in the IPBES conceptual framework. Source: Own representation.



Agriculture is the most important anthropogenic activity responsible for terrestrial biotic resource commodities, producing 2121.6 million tons of grain, 391.6 million tons of oilseed and 120.5 million tons of cotton globally in 2008 (USDA, 2009; UNEP 2010). Wood harvesting, generally associated with tropical and subtropical regions, is another important activity for terrestrial biotic resource production, accounting for

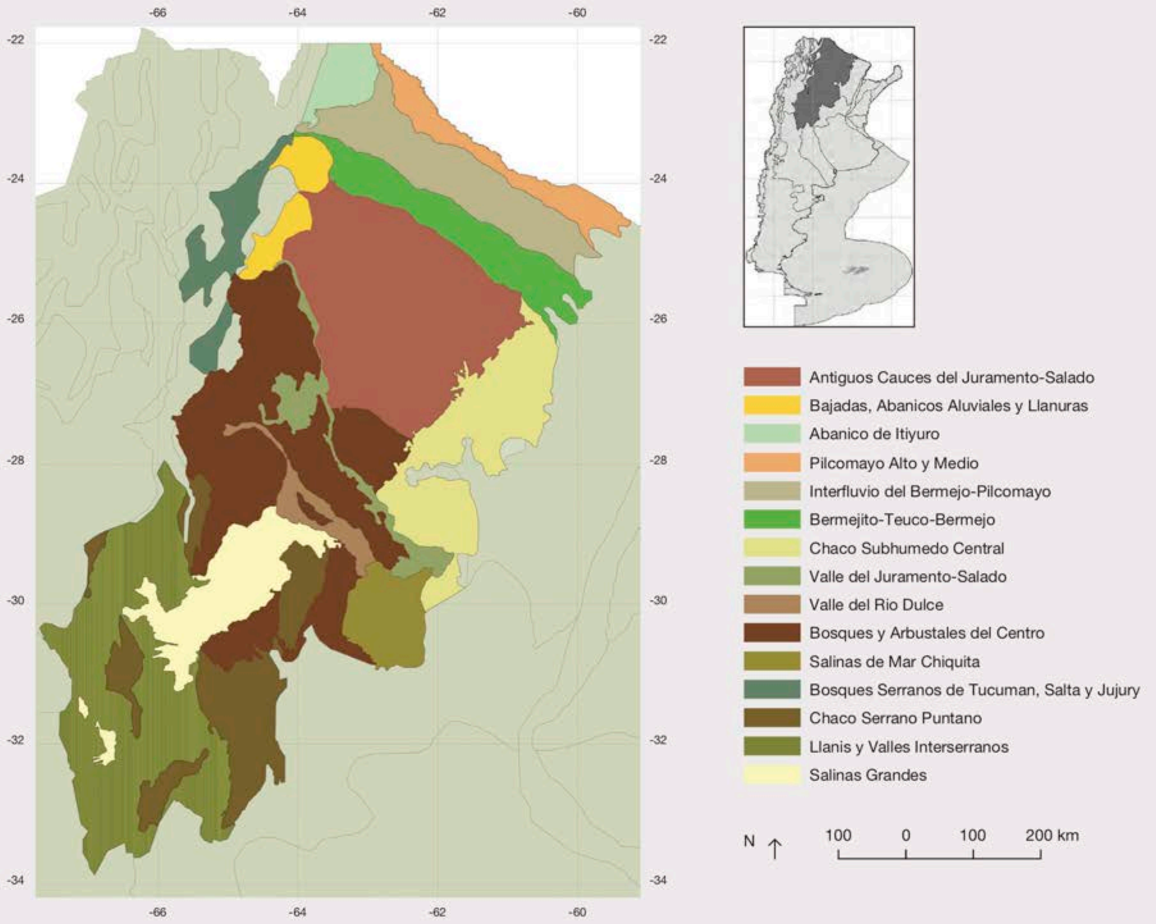
1.55 billion m^3 of wood annually (FAO, 2009). Other activities implying significant terrestrial biotic resource extraction include grazing and energy production, which are relatively smaller compared to the two previous categories. In addition, relatively insignificant amounts of terrestrial biotic resource are extracted through recreational sports (mainly hunting) and pharmaceutical uses.

Tropical and subtropical grasslands, savannas and shrublands are well represented in South America (Figure 5.10). The Latin America and the Caribbean region support large areas of tropical savannas and temperate grasslands. The Río de la Plata grasslands are the largest complex of temperate grasslands ecosystems in South America, covering approximately 750,000 km^2 within the Pampas of Argentina and the Campos of Uruguay, northeastern Argentina, Paraguay, Bolivia (Chaco ecoregion) and southern Brazil. The highest rates of endemism in the grasslands of the region are found in the páramo and puna systems, covering the upper parts of the tropical Andes from southern Venezuela to northern Peru (WWF, 2016).

Figure 5 10 **Map of biogeographical realms and biomes derived from WWF Terrestrial ecoregions dataset. Source: Map produced by UNEP-WCMC (2016) using data from Olson *et al.* (2001).**

Tropical grasslands have, and will continue to be under pressure to support global demand for biomass and food, resulting tropical forest and savannas conversion for this purpose. Habitat change in particular in tropical regions has been a main cause of global losses of biodiversity. One of the areas where this transformation is resulting in transformation of land use is the savannas in the Chaco Region (Figure 5.11), as result of land demand for soybean production, cotton and cattle expansion.

Figure 5 11 **Map of Chaco seco ecoregion and its ecosystem complexes. Source: Morello *et al.* (2012).**



Grasslands in general, are the units of analysis that as a whole present a rising trend in all major pressures on biodiversity: land degradation and land use change; climate change; land-based pollution; unsustainable use of natural resources and invasive alien species. Regional biodiversity declines are most dramatic in the tropics. A recent analysis by Brooks et al. (2016), using the UNEP (United Nations Environment Programme) regional and subregional classification as employed at the International Union for Conservation of Nature global red list database, found that 13,835 species occur within the Latin America and the Caribbean region, and that 12 per cent of these are threatened with extinction. In America, tropical and temperate grasslands were a good provider of "new lands", with soils rich in nutrients and good structure, and could be directly used for agriculture. Trends show a rising demand of land from these areas (UNEP, 2014). The food context is accompanied by rising demands for biofuels, biomaterials and biomass that compete among others with food supply. Changing diets in the national and international context, produce trade-offs on the regional and local level and models of agriculture production.

Native grasslands and savannas formerly occupied truly immense areas of the Americas and large areas still exist, though in varying states of ecological integrity, such as Pampas/Chaco/Espinal, Great Plains/savanna, and Rolling Plains/Cerrado. However, much of the grasslands and savannahs of the Americas have been greatly impacted, especially in North and South America. Different organizational scales are directly related to the grassland transformations. International trade and global demand for food, feed, biomass for biofuels, biomaterial and others, resulted in government policies that promote exports, which in turn are driving forces transforming lands for extensive agriculture and cattle grazing to fit the requirements of international markets. The issue is generating two syndromes that affect sustainability of grasslands: *agriculturisation* and *pampeanisation* (savannisation) (Manuel-Navarrete et al., 2005; Pengue, 2005).

Grasslands current scenario and regional analysis

During the last 20 years, significant challenges exist in any attempt to address the continued land use changes from grasslands and savannahs to agricultural systems due to spatial, economic and temporal considerations. Tropical and subtropical savannas, represented by the Chaco Region, are a good example of the deforestation expansion with focus on soybean expansion for sustaining international demand. Forest cover change monitoring in the Gran Chaco region in South America was undertaken using visual interpretation of Landsat satellite images, taken at monthly intervals throughout 2013. The Gran Chaco Americano is a region of forest habitat converted to savanna (Morello et al., 2012), with exceptional biological diversity and unique ecological processes being impacted. "It covers an area of 1,066,000 km^2 in four Latin America and the Caribbean countries; most of the region is in Argentina, followed by Bolivia, Paraguay and in smaller proportion, Brazil. Changes in land use were detected in 502,308 ha in 2013, the equivalent to a deforestation rate of 1,376 ha per day. Paraguay had the highest proportion of land use change recorded with 236,869 ha, followed by Argentina with 222,475 ha, and then Bolivia with 42,963 ha. According to the spatial distribution and trend of deforestation identified at the provincial, departmental, and municipal level, the Boqueron and Alto Paraguay departments had the highest rates of deforestation recorded around the Gran Chaco region". (UNEP-WCMC, 2016). In Argentina, deforestation is concentrated in the provinces of Santiago del Estero, Salta and Chaco; whereas in Bolivia the province with the largest area of change was Santa Cruz.

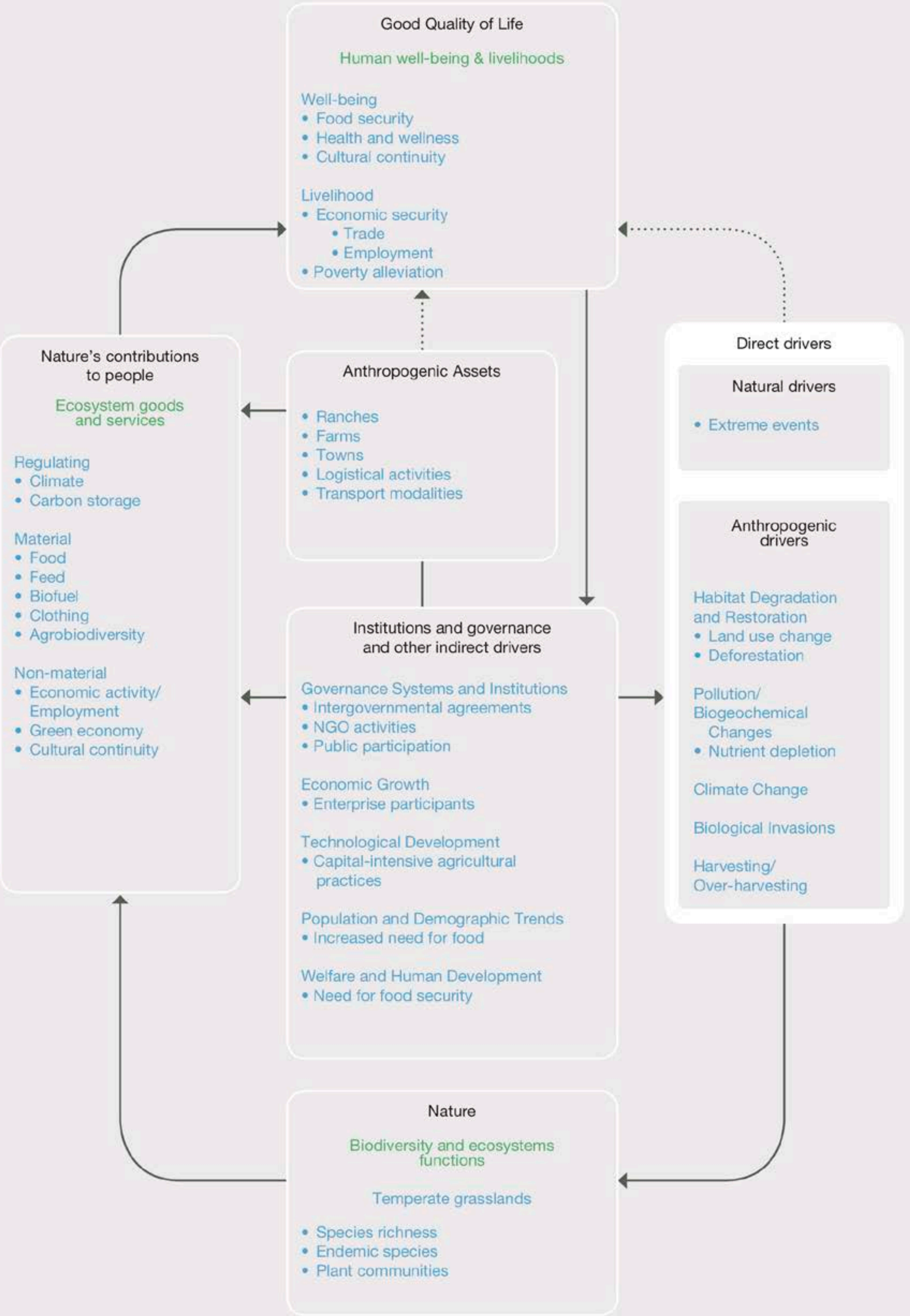
With a loss of over half a million hectares of forests in 2013, the land-use change in the Gran Chaco region is of great concern, and is primarily driven by the international demand for food, particularly meat production in Paraguay and soybean in Argentina (Caballero et al., 2013). Trade-offs in terms of land demand, rural development and national incomes are critical issues. Local or international goals could produce different results.

Main drivers are related to changing diets in western and eastern societies, China demands and the introgression of financial markets and big investors in rural communities and an expanding middle-class (UNEP, 2014) are changing the main global goal for societies: food security. On the other hand, decisive action is needed to change the present trajectory. Policies, which would limit or counter the demand of land and land use changes, particularly in developing countries, where cashcrops are seen as an opportunity to take advantage of a global demand. Agricultural intensification and expansion of arable land in tropical and subtropical grasslands for international trade will continue to expand. Latin America and the Caribbean region is regarded as second, only to sub-Saharan Africa, in terms of the potential for further arable expansion (Lambin et al., 2013), and despite droughts and water scarcity in some parts, it also holds the highest share of global renewable water resources (UNEP, 2014). Growth in sugarcane, palm oil and coffee plantations, as well as expansion of livestock production continues, often leading to deforestation, fragmentation, and overgrazing of the converted pasturelands (Michelson, 2008).

In particular, the Atlantic coastal forests, as well as tropical savannas are the most rapidly changing biomes in the region, threatened by advancing agricultural frontiers and rapidly growing cattle production (Magrin et al., 2014). This expansion and intensification of agriculture and pastureland is resulting in a decline in the area and quality of habitats and an associated increase in pollution of water courses and loss of biodiversity.

5.4.5 Temperate grasslands unit of analysis– Agricultural intensification

Figure 5 12 Temperate grasslands unit of analysis viewed in the IPBES conceptual framework. Source: own representation.

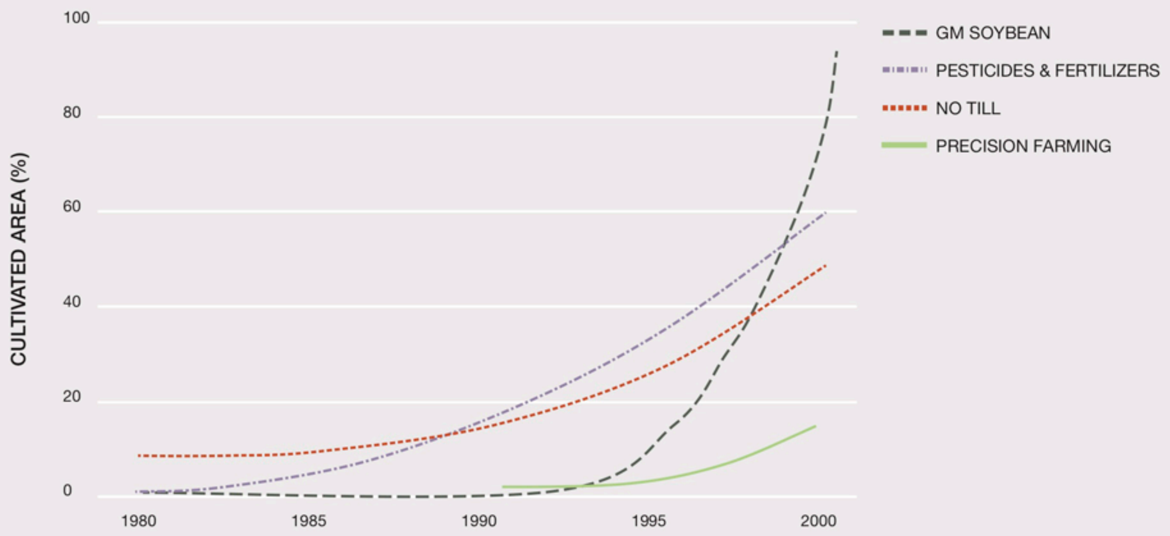


Rapid economic growth and social inequity have created certain associated pressures on the natural resources of this unit of analysis, particularly associated with the agricultural intensification. Demand for new lands and land use changes are the driving forces in the business as usual scenario. This is directly related to global trends in demand for biomass (agroindustry, biofuels and biomaterials). Conversion of grasslands to croplands is one of the key drivers in this situation. Grassland losses are significant, even in relation to other major biomes in North America. Most of the grassland loss in Canada occurred before the 1930s as a result of such conversion to cropland (UNEP, 2016b). Estimates of total loss prior to the 1990s include 97 per cent of tallgrass/savanna in southern Ontario, 70 per cent of prairie grasslands, by far the largest of Canada's grasslands, and 19 per cent of bunchgrass/sagebrush in British Columbia (Federal, Provincial, Territorial Governments of Canada, 2010). Fragmentation and land use changes is generating a degradation of natural resources and climate change, particularly where fire is used as a management tool. In Latin America and the Caribbean, the use of fire in agriculture is widespread in the region. Native forests, grasslands and other natural habitats are burned after being cleared to provide more land for agriculture; in some areas fire is also used as part of crop rotation practices. Overall, emissions from agriculture and deforestation-related fires in the region are a major contributor to atmospheric trace gases and aerosol mass concentrations (UNEP, 2016a).

Grasslands are following the fate of native forest areas. Demand for land is the driving force on the last native grassland. These changes occur in certain hotspots whose locations reflect the close and complex links between land cover, agriculture and consumption patterns both inside and outside the region (Hecht, 2014). Processes like forest clearing for creating pastures and agricultural land are still important, but have shifted from forests to other natural ecosystems, like Cerrado (Brazilian savanna) and grasslands, where soybean crops are replacing native grasslands in Argentina, Bolivia, Brazil, Paraguay, and Uruguay. Cattle production and feedlots are other main factor. In the USA, land-use scenarios assume that suburban and exurban areas will expand by 15–20 per cent between 2000 and 2050, cropland and forest areas are projected to decline compared to 1997, by 6 per cent and 7 per cent, respectively, by 2050 (Brown et al., 2014).
Several practices and policy issues are being implemented for better understanding and decision-making. Argentina recently implemented a national zoning plan (i.e., the Forest Law) to reduce further forest loss (Piquer-Rodríguez et al., 2015). For example, grasslands in Uruguay are increasingly under sustainable production systems that promote soil conservation, which is reducing land degradation (Hill & Clérici, 2013).

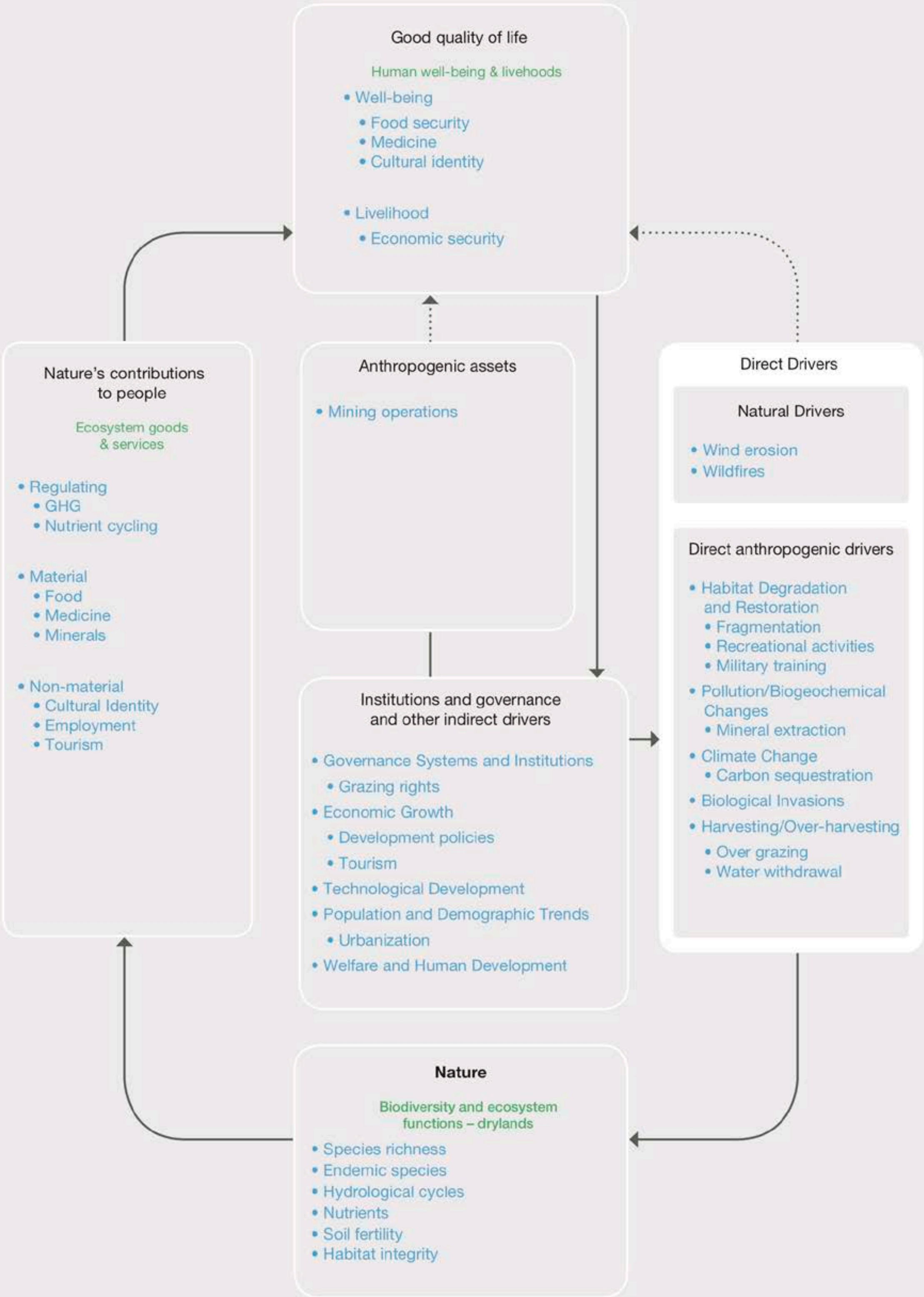
Agriculturization is a primary process in temperate grasslands with concentration in grain and crops production and displacement of cattle production to feedlots or other areas more marginal. The process has been well investigated by Gallopin et al. (2003) at the Economic Commission for Latin America and the Caribbean. New technologies play a relevant role in terms of agriculturization process on grasslands (Figure 5.13). The incorporation of modern technologies such as transgenic crops, no tillage practices, precision farming, herbicides and chemicals promote strong transformation to practically the whole of the remaining grasslands of the Americas.

Figure 5 13 **Technology incorporation for soybean production in farming systems of Argentina between 1980 and 2000. Source: Satorre (2005) and Viglizzo *et al.* (2011).**



5.4.6 Drylands and deserts unit of analysis– Exceptionally fragile diversity, resource demands, and ever-diminishing moisture

Figure 5 14 Drylands and deserts units of analysis viewed in the IPBES conceptual framework. Source: own representation.



Due to the unpredictable aridity of drylands (primarily cool and hot deserts, as well as arid and semi-arid shrubland, in Mesoamerica- and North America), both the biota and the human cultures associated with drylands have evolved a remarkable set of adaptations and cultural traditions to deal with this unpredictability (Chapter 2). Thus, despite the harsh conditions, or perhaps because of them, this biome has exceptionally high levels of biodiversity in several groups, notably plants, mammals and reptiles; there are over 30,000 plant species in the southwest USA and the State of Arizona in the USA has over 200 snake species, 2/3 the number of species in the entire Amazon (Chapter 3).

Despite water limitations, this biome provides significant provisioning services such as cattle grazing and agricultural production, though the latter is highly dependent on a non-sustainable use of irrigation via groundwater withdrawal and over allocation of surface water. However, in many cases, agricultural activities are abandoned: croplandsdue to water shortage and over grazing severely damages rangeland, resulting in the dominance of non-native species, such as *Cenchrus ciliaris* (Chapter 3). Based on the Fragmentation Index reported in (Chapter 3) only about 4% of undisturbed drylands remain, which puts it barely above the index for grasslands, one of the most heavily impacted biomes, with the main drivers being agriculture and mineral extraction. The future of drylands under climate change is unclear; temperatures may increase or stay the same.

Climate change forecasts indicate an increase in temperature, but no clear trend in annual precipitation in drylands in North America, although timing of events is likely to shift (Cook & Seager, 2013). As a consequence, potential evapotranspiration and drought severity will increase in dryland regions. Drought conditions are already common in the desert southwest, and drought periods are expected to become more frequent, intense, and longer (Garfin et al., 2014). The consequences for biodiversity are not entirely established, although drought results in a large decline in plant cover and richness, which likely impacts wildlife populations (e.g. Mulhouse et al., 2017). However, some predictions indicate that desert ecology will be impacted, resulting in perhaps half of the bird, mammal and butterfly species in the Chihuahuan Desert being replaced by other species by 2055 (Chapter 2). Drought also reduces free surface water, a resource already severely limited in most dryland regions and this reduction will affect wildlife. For instance, drought impacts desert reptiles because there is less free water for them and their prey. As many reptiles rely on their diets to obtain water if they cannot drink free water, they may die from dessication if they cannot eat enough (Schmidt-Nielsen, 1997). Drought has even more severe consequences for amphibians, as most require free water in which to live and reproduce.

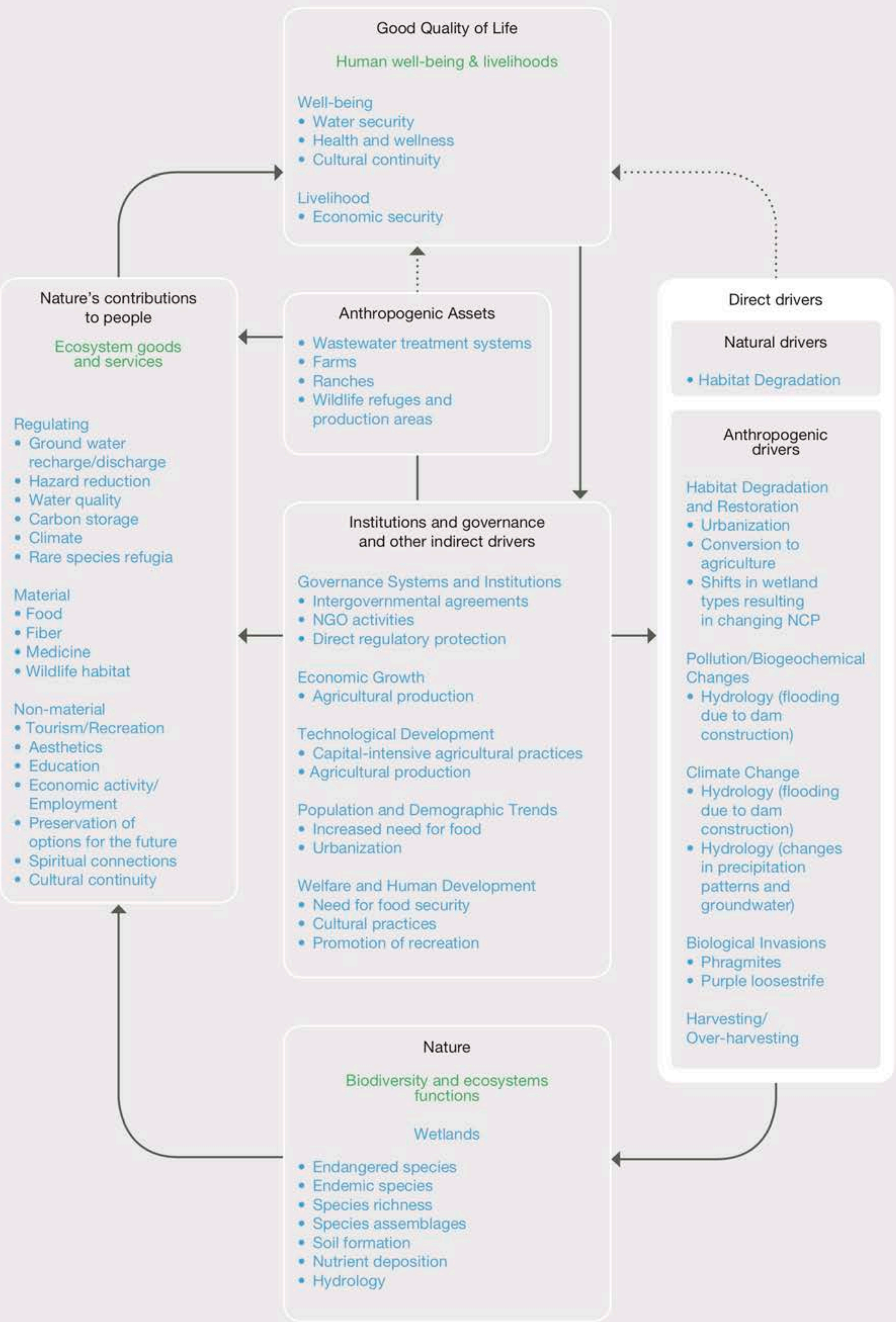
Although arid land vegetation tends to show high resilience to climatic fluctuations, currently the driest part of the loma desert vegetation appears to be at a tipping point. According to the fifth IPCC report, this area of the desert, and northward, is predicted to experience higher temperatures, but possibly more precipitation over this century, whereas more southerly parts of the desert are predicted to experience increased temperature and decreased precipitation. Increased rainfall could eventually detain present loma dieback. However, the southern end of South American desert is expected to dry further, in which case its vegetation could follow a similar trajectory today seen in the more northerly lomas. Overall, climate change and rampant development in coastal areas of Chile could become major threats to endemic western dryland biodiversity. Currently 35% of Chilean table grapes are grown in the southern part of the desert biome and its transition to the Mediterranean-climate area in Chile (ODEPA, 2013). Given expected increasing water scarcity in an increasing arid climate, grape-growing activity is likely to further affect terrestrial and aquatic biodiversity.

Although fire is far less prevalent in Caatinga than in adjacent Amazonian forest and Cerrado (de Araújo et al., 2012), fire frequency could increase with increasing aridity, predicted by the fifth IPCC report. This, however, will depend upon how woody cover evolves taking into account that vegetation response of Caatinga to precipitation tends to be nonlinear (Souza et al., 2016) and that a carbon dioxide fertilizing effect is possible. That Caatinga lies adjacent to wetter biomes is positive for providing habitat suitability elsewhere under climate change (c.f. Oliveira & Cassemiro, 2013).

As with the Tundra, climate change is the major threat to drylands, though urbanization is also a serious, continuing threat. Drylands would be expected to continue to be impacted by changing climate under the Fortress World and Market Forces archetype. While improvements with respect to climate change can be expected under the Policy Reform archetype, it is likely that scenarios that can be classified under the Great Transition archetype are the only ones that could reverse current trends.

5.4.7 Wetlands – Policy potentialities

Figure 5 15 Wetlands viewed in the IPBES conceptual framework. Source: own representation.



Wetlands constitute one of the more ubiquitous types of ecosystems throughout the Americas, providing a wide range of NCP and occur as a significant component within the following units of analysis: temperate and boreal forests, montane systems, grasslands, tundra, freshwater surface waters and water bodies, coastal habitats, and production systems. Although scattered across these units, wetlands have the shared charateristic that they are areas where the soil is saturated at a frequency and duration such that the soils are physically and chemically modified to form “hydric soils” (e.g. peat) and the vegetation is dominated by plant species adapted to growing in saturated conditions; such species are referred to as “hydrophytes” (e.g. cattails (*Typha* spp.)) (Laboratory, 1987). Wetlands may be characterized by standing water throughout the year (e.g. marshes), or water may never be visible at the surface of the ground, though saturation is close enough to the surface as to affect the soils and influence the plant community (e.g. some temperate swamps). Thus, wetlands are transitional between purely aquatic ecosystems and purely terrestrial ecosystems.

Wetlands are recognized as providing the full range of ecosystem goods and services defined in this assessment (Figure 5.15). For example, they provide provisioning services, such as food in the form of waterfowl, seafood, and cultivated rice (*Oryza sativa* and *O. glaberrima*); regulating services in the form of groundwater recharge and discharge zones, shoreline protection, as well as contaminant removal; and cultural services, such as aesthetic enjoyment, recreation, and are important culturally, such as the role of wild rice (*Zizania palustris*) in the culture of some Native North Americans (Mitsch & Gosselink, 2007; Vennum, 1988).

As noted above, wetlands occur as a significant component in seven of the 17 units of analysis recognized in this assessment. Indeed, they occur to at least some extent in all of the units, except deep water habitats. The importance of wetlands is amply demonstrated in terms of NCP by Figure 5.15.

On a worldwide basis, 64% of the wetlands that existed in 1900 have disappeared (Davidson, 2014; Ramsar, 2006). The reasons for this decrease are varied, but are primarily due to changes in land use, with the majority of wetland loss attributable to conversion to agriculture and forestry (Poulin et al., 2016). Ramsar has monitored 1,000 sites since 1970 and has found that wetland loss continues, with these sites shrinking by an average of 40% by 2008. This loss of wetland is not distributed uniformly on a global basis. Dixon et al. (2016) found that for Oceania, North America and Africa, the rate of wetland loss has substantially decreased. However, rates of loss for Asia and Europe continue fairly unabated.

For North America, the reduction in the rate of loss has been accomplished primarily through policy intervention. The USA Federal Government has enacted laws and regulations protecting wetlands, as well as encouraging conservation measures through government programs, and in the non-Governmental organization sector. While Canada has no specific Federal legislation protecting wetlands (Environment Canada, 2016), it does have a national policy of wetland conservation on Federal lands (Canada, 1991). However, wetland protection is provided indirectly at the national level through a variety of laws and regulations including, Canada Wildlife Act, Fisheries Act, Migratory Birds Convention Act, Species at Risk Act, and Canadian Environmental Assessment Act. Additionally, Canadian provinces have enacted a variety of laws intended to conserve wetlands (Rubec & Hanson, 2009).
To assess the effectiveness of these measures specifically in the USA, the federal government began monitoring the extent and type of wetlands in the coterminous USA in 1970, as well as the quality of the wetlands more recently (Dahl, 2011; USEPA, 2011). Dahl (2011) reported that the rate of wetland loss in the USA has decreased from an annual loss of 185,425ha in 1950-1970 to 5,590ha in 2004-2009; a decrease of 97%. In fact, in the period of 1998-2004, there was an actual net gain in wetlands in the USA of 12,955ha per year.

In addition to the work by Dahl (2011), Dixon et al. (2016) has evaluated wetland trends for all of North America and found that 4% of inland wetlands were lost in the period 1970-2008, while 28% of coastal/marine wetlands were lost, for an overall loss of 17% for the two classes of natural (not manmade) wetlands. It is notable that the 4% loss of inland wetlands in North America compares to 31%, 39%, 59%

loss of inland wetlands in Africa, Asia, and Europe, respectively. On the other hand, Poulin et al. (2016) have assessed the effectiveness of recently established provincial policy in Quebec. They found that despite legislative mandates for wetland mitigation, nearly all wetlands subject to permit agreements were lost without commensurate wetland mitigation (though other forms of mitigation did come into play, such as upland preservation).

While these figures argue for the effectiveness of policy efforts, they mask other underlying considerations.

The capability of any ecosystem to delivery its characteristic suite of goods and services is dependent on the integrity of the structure and function of the ecosystem. While Dahl (2011) reports that for the period of 2004-2009, wetland losses were statistically insignificant overall, there were quite decided shifts in wetland types. For that period, freshwater wetlands actually increased by 8,900ha, but this increase was attributable to an increase in agricultural, industrial and urban ponds. Non-forested freshwater wetlands (which were considered in the report to be the ones expected to have a reasonable degree of ecological integrity) actually decreased by 72,900ha, with forested wetlands decreasing by 249,200ha; while the types and level of ecosystem goods and services delivered by constructed agricultural, industrial and urban ponds are not the same as lost from the forested systems, they may nevertheless deliver more NCP for a specific service, such as food production. The tension between the valuation of different wetland ecosystems and their associated NCP is also exemplified by somewhat conflicting legislation. For example, while there is federal legislation in Canada protecting naturally occurring wetlands, there is also local legislation, such as Ontario's Tile Drainage Act that promotes drainage of wetlands for agricultural purposes (Environment Canada, 2016). A similar situation exists in the USA at the state and local levels.

It is also instructive to look at the land use changes that accounted for the shifts in wetland types during the period of 2004-2009 (Table 5.1).

Table 5.1. Changes in wetlands attributable to indicated land use classification 2004-2009. Source: Dahl (2011)**.**

Land use category	Net change in wetland area (hectares) attributable to change to indicated land use
Deep Water	-46,947
Urban Development	-24,951
Rural Development	-27,101
Silviculture	-124,429
Agriculture	40,494
Other	157,738

Conversion to silviculture accounted for the greatest decrease in wetland extent, while "Other" accounted for the greatest gain. "Other" includes land use changes that are so recent that the ultimate land use category could not be determined. However, it also included newly constructed wetlands and establishment of conservation easements. Thus, it is apparent from Table 5.1, that wetland loss continues with respect to underlying causes. These causes also point to other factors contributing to wetland loss. The conversion to deep-water habitats is largely from salt marsh loss resulting from wave action encroachment allowed by fragmentation of salt marsh associated with oil and gas production. Similarly, urban and rural development can have synergistic effects through increased nutrient, heavy metal, and other pollutant loading to nearby wetlands.

The United States Environmental Protection Agency (USEPA, 2011) considers these latter concerns as potential threats to the quality of wetlands. For example, they list road runoff as a source of copper, lead, and vanadium contamination. Similarly, they point out that agricultural activities can be the source of heavy metals such as cadmium, copper, nickel and tin, as well increased nutrient and sediment loads to wetlands. The overall effects of these contaminants may be reflected in the fact that wetlands in areas with intense agricultural activities also tend to have lower floristic quality compared to areas with less intense agriculture.

Historically, wetland degradation near large urban centers has been particularly acute. This trend is likely to continue, given the limited options for avoiding land use conflicts in densely settled areas. Climate change is a growing threat to wetlands across North America. Across the peatlands of Canada and Greenland, climate change is likely causing widespread permafrost degradation, alterations of snow and ice regimes, and changes in ultraviolet radiation (Jeffries et al., 2013). Changes in freshwater geochemistry including eutrophication arising from the release of stored nutrients in permafrost and deepening of the active soil layer have been reported (Meltofte, 2013). In boreal peatlands, climate change is expected to trigger increased drought and so increased fire frequency and peat loss (Galatowitsch et al., 2009). Climate change projections for the prairie pothole region suggest shifts in hydrology that will make most of the region unsuitable breeding and migratory habitat for waterfowl (Galatowitsch et al., 2009; Johnson et al., 2005). Climate maladaptation by the agricultural sector, needing to secure more water sources, seems likely to result in water diversions and groundwater extraction, adversely altering wetlands in many parts of North America, including the prairie pothole and Everglades wetland landscapes (Galatowitsch et al., 2009; National Research Council, 2014).

Wetlands in seasonal tropical climates, as is the case of the Palo Verde wetland, are governed by extreme seasonal hydrologic fluctuations and are characterized by rapid vegetation responses to changes in water level. Climate change models in the seasonal Palo Verde wetlands in Costa Rica predict reduced rainfall and a drier wet season. Based on the distinctive composition of wet and dry season vegetation, and high species richness in the wet season, local loss of diversity is predicted accompanied by increased abundance of drought-tolerant emergent species (Osland et al., 2011).

Given a general tendency for increased aridity and changes in seasonal rainfall distribution in South America over the coming century, wetlands are likely to be negatively impacted by climate change (Junk, 2013). However, there are many uncertainties given regional climatic variation. For example, some climate models show increases in rainfall and in discharges of the Paraguay Basin, while others show reductions (Marengo et al., 2016).

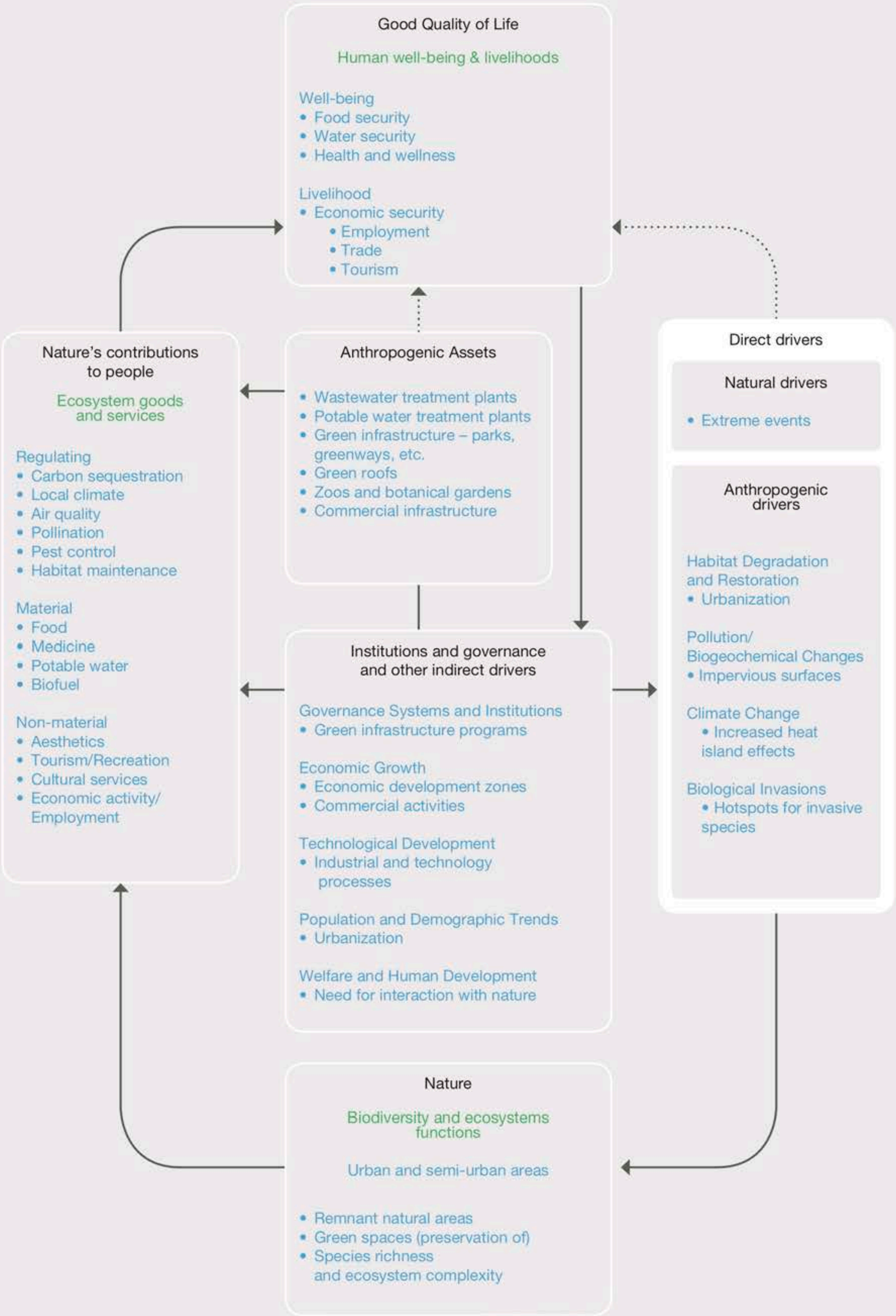
The two main drivers affecting wetlands currently and expected to continue to do so in the future (Figure 5.15) are habitat degradation and climate change. With respect to habitat degradation (i.e. primarily conversion of wetlands to agricultural use), the information presented above speaks to the feasibility and potential effectiveness of policy intervention in wetland conservation and, thus, speaks to the potential implications of the archetypes. The majority of wetland loss that has occurred in North America occurred, as the land was being settled and converted to agriculture. We see this driver still taking place in other areas of the Americas, notably South America where land is being converted to agricultural purposes, such as growing soybeans. Thus, under the Market Forces archetype, we would expect to see continued loss of wetlands in areas that do not already have protections. Under Fortress World, we would expect a similar, though likely more severe, trend as market forces and expanding populations requiring food would result in the same trend observed in North America in the 1800 to mid-1900s. The relative effectiveness of policy intervention is well-evidenced by the above discussion and thus, under the Policy Reform archetype one would expect a reduction in the rate of wetland loss where it is still prevalent, though depending on the policies, shifts among wetland types may occur as the do in USA, with concomitant shifts in the exact NCP provided. The adoption of policies, such as those in USA and Canada and the recent significant set aside of the Llanos wetlands of Bolivia, could be a significant boon to maintaining the NCP provided by wetlands. The set aside in Bolivia also points to what might happen under the New Sustainability Paradigm archetype (i.e., an archetype similar to the Policy Reform and the Great Transition group of archetypes). Despite being in a region where land use changes to agriculture is proceeding at a substantial rate; it is possible to set aside ecosystems whose NCP values are recognized.

While the Policy Reform family of archetypes hold promise with respect to addressing land use changes, it is likely a much less effective scenario for curbing wetland impacts due to climate change. Additionally, climate change may also result in increased water withdrawal from wetlands or the aquifers that supply groundwater-fed wetlands. Thus, agriculture and climate change can be viewed as synergistic drivers, and for reasons covered under Tundra and Boreal Forest, effective approaches in dealing with this synergistic pairing will require more radical approaches, consistent with the Great Transition family of scenarios.

The above focal analysis provides a good indication of the complexity of determining what “the best” use of world’s natural capital is. Multiple drivers, teleconnections, telecoupling, differing socio-economic conditions, and differences in cultures and values are all considerations in trying to create a sustainable world. Section 5.6 discusses the detailed considerations in developing specific scenarios that can inform the policy process in attaining this goal.

5.4.8 Urban/Semi-urban – Effects on multiple aspects of human well-being

Figure 5 16 Urban/Semi-urban viewed in the IPBES conceptual framework. Source: own representation.



Urbanization will continue as world population grows and may have its greatest effect in intermediate-sized cities, which have the highest growth rates (Chapter 2). The continuation of urbanization will impact other units of analysis, such as agricultural systems and can result in significant impacts to NCP provided by those systems, such as provisioning of food (Chapter 3). For example, Schneider et al. (2012) estimates that by 2030, urban expansion in the Midwest of the USA may reduce agricultural land that could feed up to 532,000 people.

While urban centers are impoverished ecological systems relative to many ex-urban areas (including agricultural systems and the landscapes within which they are imbedded (Chapter 3), they still host a variety of species and underpin a variety of ecosystem services, especially with respect to regulating and cultural services (Chapter 2).

Some urban areas, such as the City of Detroit, Michigan, USA, park systems contain remnant tracts of vegetation that are only slightly changed from pre-settlement times due to the fact that they were parts of estates before urbanization spread to their area and were protected as part of park systems (Weatherbee & Klatt, 2004). Indeed, one of the natural communities (Mesic Flatwoods) recognized in Michigan, was first described just a few years ago based on the urban park Belle Isle, located in the Detroit River, between the downtowns of Detroit and Windsor, Ontario, Canada (Cohen et al., 2015). These observations argue for continued inventorying of the biological assets in urban areas, even in areas that are considered highly urbanized and studied (Chapter 3).

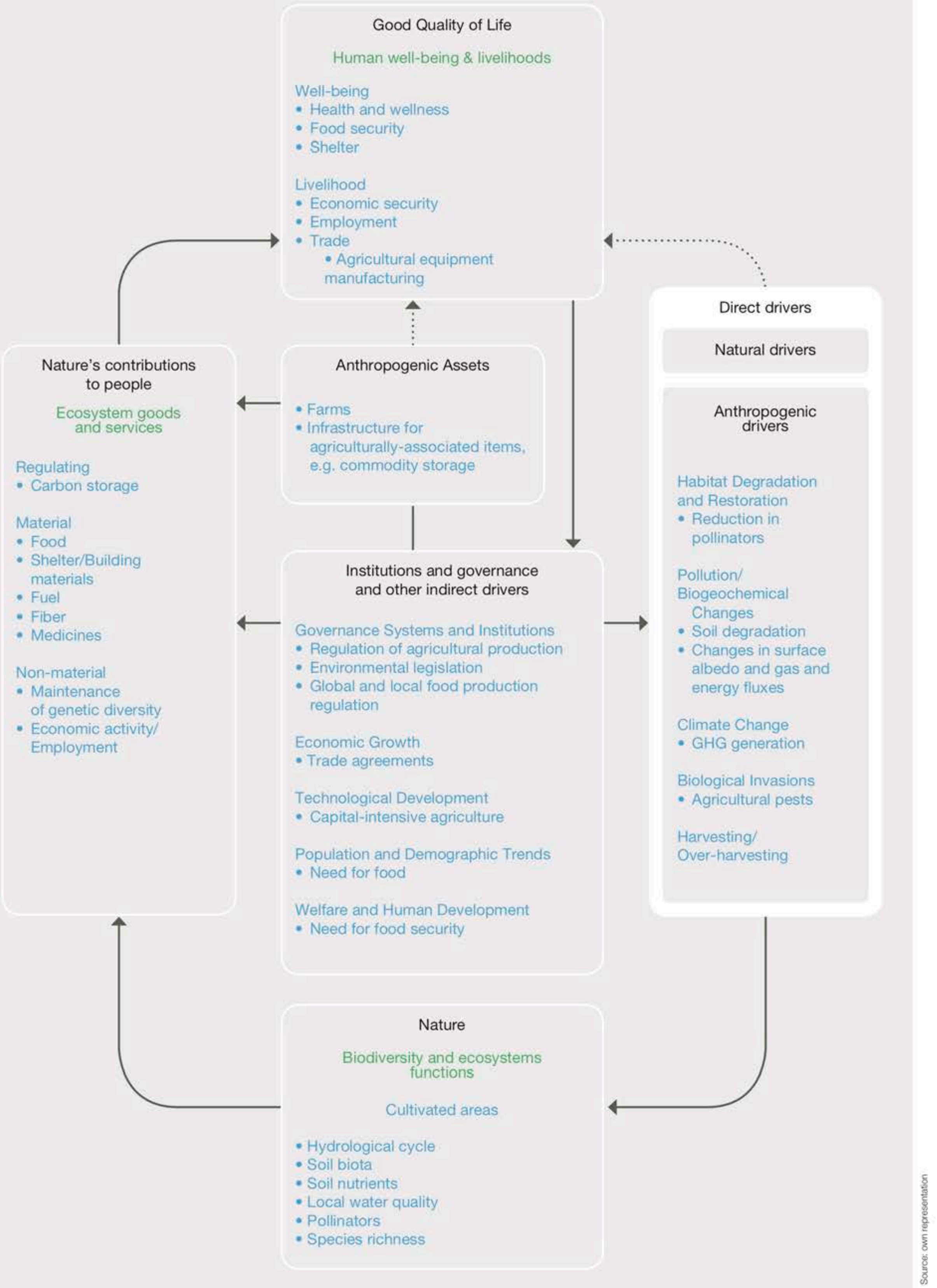
Perhaps the greatest impact on biodiversity due to urbanization may be indirect, through the continued reduction of human-nature interactions, which have been shown to be beneficial to people in general and even utilized in human medicine as an adjunct to cancer treatment (Chapter 2) (Cimprich & Ronis, 2003; Louv, 2008). The disconnect from nature is likely to result in disaffection toward nature and reduced motivation to protect, due to a lack of understanding.
It is in the area of urban planning that some of the greatest opportunities for meeting the Sustainable Development Goals (SDG) by employing technological advancement in preserving and enhancing the function of urban ecosystem and mitigating the negative consequences of urbanization exist. For example, the use of designed wetlands for the treatment of storm runoff and sanitary wastewater can lower point pollution of surface water, provide wildlife habitat, and afford cultural opportunities to enjoy nature. More study is needed to determine adequate amounts of greenspace for human well-being from a variety of perspectives, Shanahan et al. (2016) have shown significant effects with 30 minutes per week of exposure to natural surroundings. There is strong evidence for a positive effect of the number of urban greenspaces on biodiversity; a relationship well established on the principle of MacArthur and Wilson's Theory of Island Biogeography (Macarthur & Wilson, 1967). Indeed, the mathematical relationship between available habitat and species diversity has been described for a number of systems.

As with most human endeavours, such as the development of agriculture and technological advances, urbanization has both significant benefits and costs. Urbanization is associated with increases in quality of life in terms of food availability, sanitation, and healthcare; it is also associated with environmental degradation, poverty, unemployment, and violence. It will take public discourse and development of sustainable development policies to insure maximization of benefits and minimization of costs.

As urbanization is one of the main causes of land use changes, reduction of the effect of this driver in urban areas themselves will require concerted effort in land use planning. Thus, various approaches within the Policy Reform and Great Transition archetypes hold promise for biodiversity conservation with respect to urbanization.

5.4.9 Cultivated areas (including cropping, intensive livestock farming, etc.)

Figure 5 17 Cultivated areas (incl. cropping, intensive livestock farming, etc.) viewed in the IPBES conceptual framework. Source. own representation.

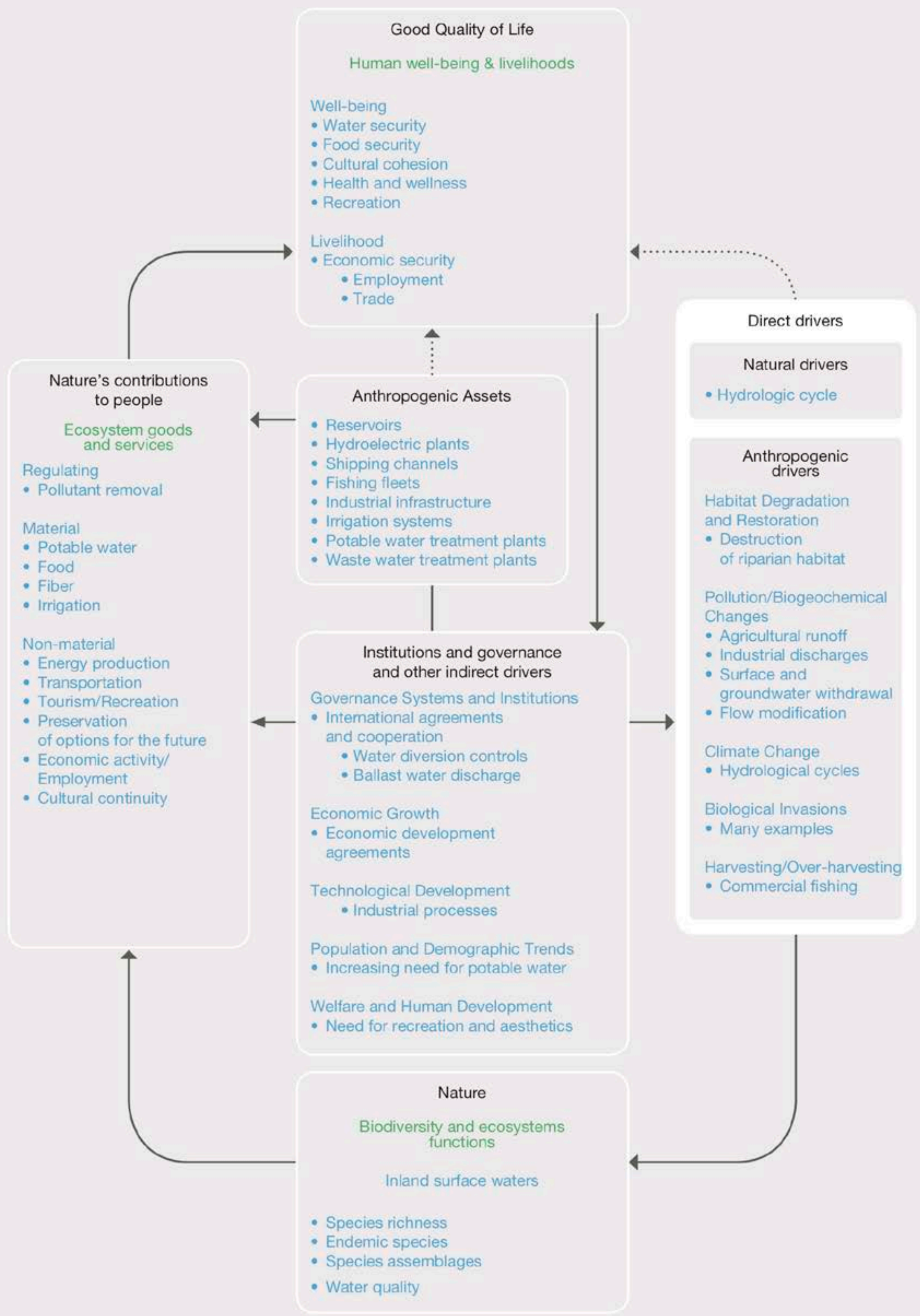


The agricultural land in Latin America and the Caribbean showed one of the larger expansions in the past 50 years (Martinelli, 2012). The challenge the region faces is to meet the large potential for food, fiber and fuel production aligned with conservation of one of the larger and unique collections of biodiversity, on the planet. Most of the increase in production was associated to the expansion of extensive agriculture over forests and natural ecosystems areas (Willaarts et al., 2014). In Brazil, one of the larger agricultural commodity producers in the regions, circa 20% of the Amazon rain forest and 50% of the dry forest (Cerrado) was lost due to the expansion of agriculture in the past 40 years (Aguiar et al., 2012; Bustamante et al., 2012). Also, pressures over the Chaco area (Bolivia, Paraguay and Argentina) due to increase of grain and beef production is critical. Latin America and the Caribbean has a key role in the international agriculture products market, as a leading exporter and producer of soybean, sugar, coffee, fruits, poultry, beef and bio-ethanol (Martinelli, 2012). The greenhouse gases emissions portfolio in the region is strongly centered in process of land cover change (deforestation, forest degradation, land degradation) (Aguiar et al., 2012) and land use by agriculture and cattle ranching. According to Sy et al. (2015), analyzing the 2010 global remote sensing survey of the FAO - Global Forest Resources Assessment, pasture was responsible for more the 70% deforestation in Northern Argentina, Western Paraguay, and eastern portion of the Brazilian Amazon (the arc of deforestation), whilst deforestation driven by commercial cropland (12-14%) had an increased pattern in time, and the hotspots found in Brazil (south western Amazon), Northern Argentina, Eastern Paraguay and Central Bolivia. In Brazil, Argentina and Mexico agriculture has already surpassed the emissions derived from deforestation (UNFCCC). Broader data published by (Graesser et al., 2015) indicate that, for the entire Latin American region, 17% and 57% of forest replacement was due to new cropland and new pastureland.

The agricultural expansion and production varies strongly in the region, and production has distinct level of cropping efficiency and intensity in different countries and biomes. Thus, intensification and extensification processes have driven the agriculture expansion in the region in the past decades. Most commoditized agriculture is highly technological and is related to private and commercial companies, but small holder agriculture plays a critical role on food production at local and regional scale (Boillat et al., 2017). Land tenure and demography in the region also play a role in the dynamic of land use change processes. The demographic configuration of the Latin America and the Caribbean region has low population density in the rural area and one of the most urbanized regions on the planet (e.g. almost 80% of the population lives in cities) (UNEP, 2014). Land tenure is a critical issue. In Mexico, Bonilla-Moheno et al. (2013) showed differences in woody cover, in natural vegetation landscape units, from common-pool systems of land tenure, in contrast to communal and private regimes, where the latter ameliorate, reducing the deforestation process.

5.4.10 Inland surface waters and water bodies/freshwater unit of analysis- The case of multiple demands/multiple drivers on natural capital

Figure 5 18 Inland surface waters and water bodies/freshwater unit of analysis viewed in the IPBES conceptual framework. Source: own representation.



Water is fundamental to all living things, the chemistry of life occurs in aqueous solution. Whether an organism occurs in terrestrial, sub-terrestrial, marine or freshwater environments it is dependent on water. Thus, all of biodiversity, as well as the NCP stemming from that diversity, link to water. While marine systems dominate the globe in areal extent, human well-being is, arguably, more closely linked to freshwater, if for no other reason than the human need for drinking water.

The distribution of water is heterogeneous, as is the specific need for water. The demands on freshwater systems are large and extremely diverse. For example, though both are areas of high intensity agriculture, the need for irrigation in the Upper Midwest of the USA is much lower than for the central valley of California. Ironically, in the Upper Midwest where rainfall tends to be adequate, 20% of the world's freshwater is found in the Great Lakes. Thus, there can be major disconnects between need and occurrence of this natural capital.

Sustainable Development Goal 6 is "Ensure access to water and sanitation for all." Chapter 2 makes clear the NCP of freshwater systems and are presented Figure 5.18. Indeed, the criticality of water as a resource, in terms of sustainability, economic activity (including as a source of jobs), and human health have been emphasized, respectively, in the last three World Water Reports (WWAP, United Nations World Water Assessment Programme, 2015, 2016, 2017). While Chapter 3 describes both discouraging and encouraging trends, the challenges facing this unit of analysis are made clear by the discussion of drivers in Chapter 4.

Two primary drivers that act synergistically are demographics and agriculture. It is expected that water use will continue to rise both absolutely and on a per capita basis due to increasing populations throughout the Americas and agricultural intensification, respectively. Though irrigation technology has improved via such aspects as in-field moisture sensors, the adoption of these technologies is slow (WWAP, United Nations World Water Assessment Programme, 2015). As agriculture intensifies, especially in South and Mesoamerica, increased pressure will be placed on freshwater systems due to water withdrawal and eutrophication due to nutrient-laden runoff. Though point-source pollution has been much reduced in North America, the same does not apply regarding non-point source pollution and agriculturally-related nutrient inputs are a major concern in the Mississippi River basin and western Lake Erie of the Great Lakes. The aspect of water withdrawal is especially troubling in Mesoamerica where, in certain areas, a third of aquifers are already over-allocated. But water withdrawal is also a serious problem as well as in North America where there is a dependence on "fossilized water" (aquifers that are not being replenished) for irrigation. The combination of the need for drinking water and irrigation is particularly problematic in the southwest USA where up to 76% of river flows are withdrawn annually (the Colorado River frequently does not reach the Sea of Cortez and its delta is 10% of what it used to be).

Linked drivers, including urbanization and energy needs, provide a challenge to freshwater systems and simultaneously meeting SDG 7 (sustainable energy) and 15 (eliminate biodiversity loss). There is no doubt that energy production via burning of fossil fuels has significant environmental consequences and that sustainable energy sources are needed, especially if urban energy needs are to be met. However, the three main current sources of sustainable energy, namely solar, wind and hydro, all come with their own ecological footprint. Hydropower is a source being widely considered in South America and there are currently a number of dams either under construction or being planned. While these will provide reliable energy, they also come with an environmental price including disruption of fish migration routes, increased sediment deposition upstream, channel scouring downstream, and disruption (increased and decreased) of annual flooding of riparian terraces traditionally used for agriculture.

While freshwater systems are undeniably an important resource to humans as drinking water, freshwater is also critical to the biological resources found in lakes, streams, and rivers. The Americas are exceptional in their freshwater resources. For example, as noted in Chapter 3, the Americas contribute 47% of the freshwater that flows to the oceans and the freshwater of the Americas is home to over 5,000 species of fish, which provide subsistence food, commercial food, and sport opportunities. However, these and other freshwater biological resources in the Americas are threatened by habitat degradation (e.g. construction of dams for hydroelectric power), climate change, pollution (as in the water quality issues for Lake Erie discussed above), and invasive species (e.g. Asian carp and zebra mussels in North America) resulting in higher extinction rates than for most terrestrial biomes (Dove, 2009; Chapter 3).

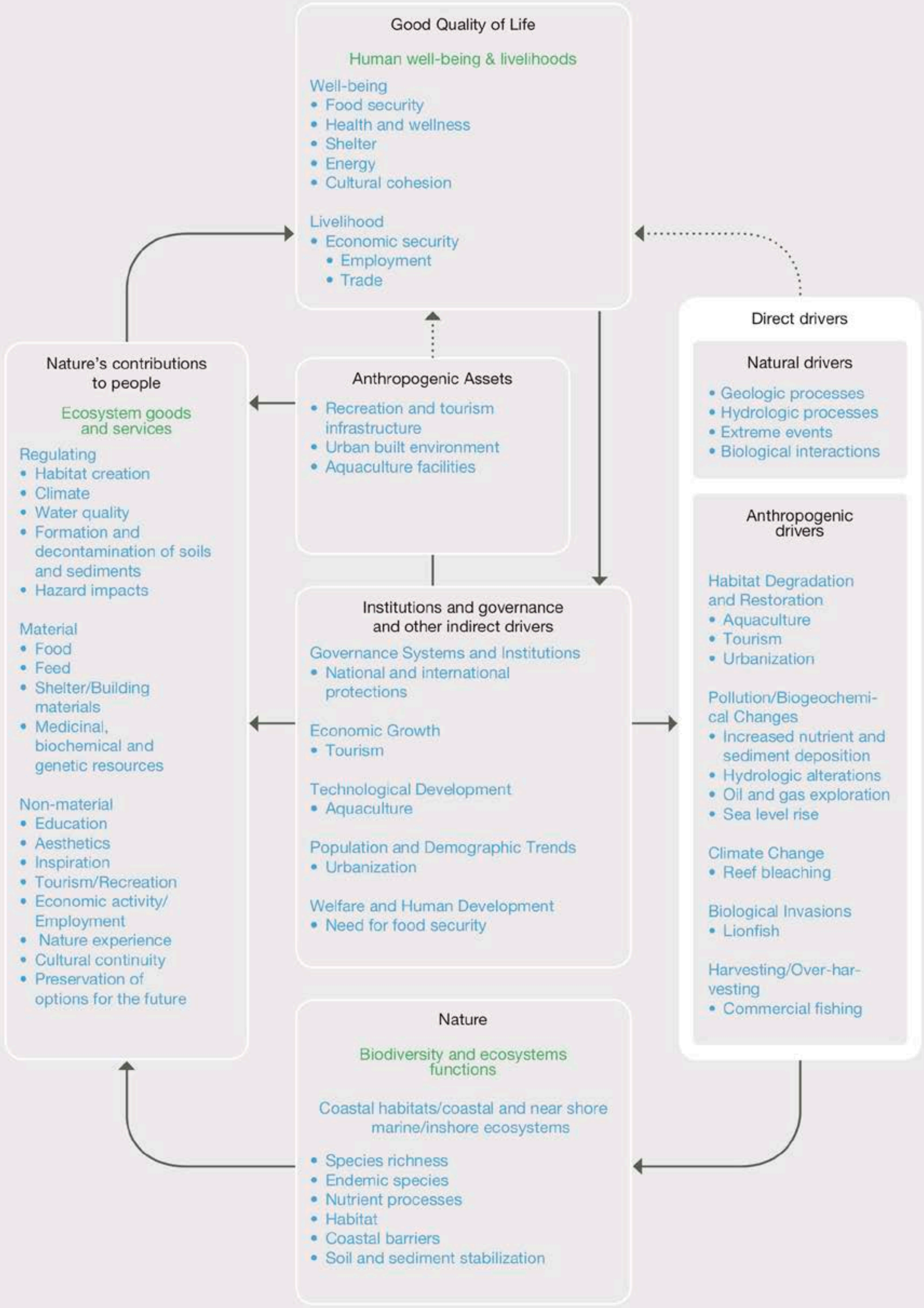
These drivers will continue to present recurring and likely increasing, challenges to freshwater resources as we approach 2050. While serious threats exist to the Americas' freshwater systems, there is also evidence that planning and international cooperation in addressing these threats through policies and intergovernmental agreements have helped some freshwater systems, notably the Laurentian Great Lakes in North America. Coordinated water pollution control by Canada and USA, and the formation of the International Joint Commission on the Great Lakes have achieved substantial levels of success in protecting the Great Lakes with respect to water removals and diversions (International Joint Commission, 2016) and reductions in petroleum, pesticides, heavy metals, and nutrient pollution since the 1970s (Hartig et al., 2009). For example, water clarity has vastly increased in Lakes Michigan and Huron, phosphorous levels have been reduced to the extent that they are now considered a limiting nutrient in the lakes, chloride levels in Lakes Huron, Erie, and Ontario have decreased (reversing a 150-year trend of increasing levels).

These improvements are credited with recovery of a number of biological resources, including bald eagles (*Haliaeetus leucocephalus*), peregrine falcons (*Falco peregrinus*), lake sturgeon (*Acipenser fulvescens*), lake white fish (*Coregonus clupeaformis),* walleye (*Sander vitreus*), and burrowing mayflies (*Hexagenia* spp.)(an important prey item in fish diets) (Hartig et al., 2009). While improvements have been noted in these measures, other pollutants, such as silica and nitrogen, have increased (Binding et al., 2015; Chapra et al., 2009; Dove, 2009; Dove & Chapra, 2015).

Thus, while policies and international cooperation has been helpful in North America, it is clear that futures that include scenarios from the Fortress World or Market Forces archetypes will not be enough to stem the increasing pressures of non-point pollution, climate change, and invasive species even at the subregion. True paradigm shifts will be required throughout the Americas to address impacts to freshwater, especially in terms of water quality and availability, in the face of increasing reliance on pesticides, fertilizers, and irrigation in agriculture in response to increasing populations and climate change. It is clear that to make progress towards the Aichi targets and SDG, serious consideration should be given to devising scenarios designed within the Policy Reform and Great Transition archetypes.

5.4.11 Coastal habitats/coastal and near shore marine/inshore ecosystems unit of analysis

Figure 5 19 Coastal habitats/coastal and near shore marine/inshore ecosystmes unit of analysis viewed in the IPBES conceptual framework. Source: own representation.



Coral Reefs. According to Knowlton (2001) the combination of nutrification, global warming, and loss of top members of the food chain (and introduced chemicals) is unprecedented over the last 65 million years. Bozec et al. (2016) concluded that reduced fishing for parrotfish and other herbivores would make reefs more resilient to warming and ocean acidification. Global warming is placing Caribbean coastal ecosystems under further stress. Predicted increased severity of hurricanes and greater rainfall seasonality for the region are also likely to increase stress (Fish et al., 2009). According to the IPCC fifth assessment report, under 4°C warming, widespread coral reef mortality is expected with significant impacts on coral reef ecosystems, this will imply a high risk of extensive loss of biodiversity with concomitant loss of ecosystem services (CB Field et al., 2014).

Mangroves. These wetland systems occur along coastal areas from the subtropics in North America to the tropical and subtropical regions of Central and South America. Like most wetlands, they provide a range of ecosystem goods and services. They provide provisioning services in the form of food production (Engle, 2011); regulating services in the form of storm protection, coastal protection, and erosion control (Anthony & Gratiot, 2012a; Marois & Mitsch, 2015; Zhang et al., 2012) and cultural services in the form of recreation and aesthetic enjoyment (Mitsch & Gosselink, 2007) (Figure 5.19). Indeed, they are considered some of the most productive wetlands on Earth from the standpoint of providing habitat for fisheries and wildlife.
On a global basis, it is estimated that over 60% of the world's wetlands have been lost and this is largely due to land use changes, primarily conversion to agricultural systems (Ramsar, 2006); these losses are not uniformly distributed among wetland types or geographic areas, but the losses continue. Between 1980 and 2007, 25-35% of the world's mangrove forests were lost (FAO, 2017a; Inniss & Simcock, 2016; MEA, 2005). Moreover, Marois and Mitsch (2015) state that the majority of remaining mangrove forests are located within 25km of major urban centers. Recent figures indicate that the loss of mangrove forest has continued with an additional loss of 1-2% per year, though higher rates occur in some regions. It is notable for this assessment that from 1980 – 2007, there has been a loss 24-28% of the areal extent of mangroves in the Caribbean with much of the loss due to conversion to urbanization, fuel wood, solid waste disposal, and aquaculture (Anthony & Gratiot, 2012b; Inniss & Simcock, 2016).

With anticipated rises in sea level and more and more intense storm events associated with climate change, this loss of mangrove forest is of concern due to their role in storm surge attenuation, shoreline protection, and soil erosion prevention. The attenuation of normal wave energy by mangroves is well known. However, it has become increasingly recognized that mangrove forests may play a significant role in ameliorating the effects of severe storm and tsunami-generated waves. Danielsen (2005) reported that villages that had a mangrove barrier suffered relatively fewer deaths from the Indian Ocean tsunami than villages without such a barrier. While some have questioned the efficacy of mangroves in the case of tsunamis, Zhang et al. (2012) have convincingly demonstrated the protective value of mangroves in the case of hurricane Wilma that had landfall in southwest Florida USA. They showed that a 7-8km wide mangrove forest reduced inundation by 80%, thus protecting inland freshwater wetlands from saltwater encroachment.
Mangroves also play a role in prevention of soil erosion. Along the northern coast of South America, sediment-laden waters from the Amazon River form extensive areas of shifting mud flats. These mud flats extend thousands of kilometers along the coast and are stabilized by mangroves. However, in some areas, the mangroves have been removed for development, or dikes built to establish aquaculture operations, isolating the mangroves. In those areas, the protective stabilization provided by the mangroves is no longer there, resulting in erosion of the mud flats and conversion of the shore to sand. The sandy soils do not support vegetation and are highly erodible requiring local communities to install expensive shoreline armoring, such as rip-rap or concrete break walls (Anthony & Gratiot, 2012b).

Conservation of the mangroves in an area may also have synergistic effects. Engle (2011), reviewed the available information on ecosystem services associated with wetlands in the Gulf of Mexico, including shrimp production. Juvenile shrimp develop in coastal wetlands, primarily marshes. However, as in the case of mangrove forests, there is an on-going loss of coastal marsh in the Gulf of Mexico primarily resulting from changes in flow patterns induced by oil and gas exploration (Rangoonwala et al., 2016). As shrimp habitat decreases, it has been found that juvenile shrimp use other coastal wetlands, such as open bays and seagrass areas. Thus, there may be ancillary benefits to the shrimp industry from mangrove conservation by providing alternative habitat for juvenile shrimp.

Despite efforts to restore mangroves in some areas in the Americas (http://www.mangroverestoration.com/), expansion of aquaculture and will likely continue to reduce the extent of this valuable ecosystem. Alongi (2002) predicted that over the next 25 years, unrestricted tree felling, aquaculture, and overexploitation of fisheries will be the greatest threats worldwide, with lesser problems being alteration of hydrology, pollution and global warming. In contrast, Ellison and Farnsworth (1996) felt that climate change would likely cause fringing mangroves to vanish. However, in recent years, mangroves have been spreading northward in Florida, expanding their range in response to warming (Cavanaugh et al., 2014). Since they are not likely to be harvested for wood or removed for aquaculture, this northward move may counterbalance some of the threats. In the Caribbean, rising sea levels will likely have a large impact on coastal areas, although mangroves have been shown to keep pace with sea level rise in some areas of the Caribbean such as Belize (McKeeand Feller, 2007).

Although mangroves provide various NCP, undeniably contributing to human well-being by reducing fatalities associated with extreme events, the drivers resulting in the loss of mangroves also contribute to human well-being; thus we have to consider the full range of consequences involved. Conversion of mangrove forests for agriculture or aquaculture contributes to food supply, urbanization may result in the general increase of the standard of living of those in the urban areas. So too, all of these drivers are associated with economic activity of one sort or another and may contribute to alleviation of poverty. Thus, various considerations need to be taken into account when evaluating the sustainable use of mangroves. Datta et al. (2012) present an approach that can help to resolve these questions of both negative and positive consequences. They review the results of a number of community-based mangrove management efforts and provide a number of observations regarding factors that contribute to the success of such efforts, such as ensuring the voices of the those depending on the mangroves for subsistence are heard and that the benefits derived from the management efforts, including the economic benefits, are distributed equally regardless of socio-economic status of the recipients.

Clearly, the situation and necessary considerations in the case of mangroves, and the NCP they supply, differ substantially from the issues with Tundra wetlands. In the case of mangroves, the drivers are both direct and indirect, and while some, such as climate change (which causing some latitudinal change northward, (Inniss & Simcock, 2016)) are global, others, such as land use change are very local. So too, there are costs and benefits in terms of NCP that are related to the relevant drivers (e.g. aquaculture provides food and economic activity). There is also clear evidence that local populations can have a direct effect on the resource, including the NCP that it supplies.

Thus, the scenario archetypes have slightly different implications in this case and pathways to a sustainable future are possibly more flexible. Under the Fortress World archetype, it is still likely that mangroves in the Americas will continue to suffer losses, though an extreme acceleration of impacts, as would be anticipated for Tundra wetlands is less likely, due to local recognition of the NCP of mangroves in terms of local fisheries and shoreline protection. However, this may be overbalanced by a presumed increase in urbanization or other land use changes, as cooperative agreements and existing protections in some areas may roll back.

As with the Tundra wetlands, a future under the Market Forces archetype will likely result in the continued degradation of this resource throughout the Americas. Assuming an even greater reliance on market forces, there may be an actual increase in impacts to mangroves, as the NCP most easily monetized, such as aquaculture, urbanization and coastal development, will likely increase; these being the factors most often cited in current impacts to mangroves, especially in the Caribbean.

A future under a Policy Reform archetype scenario holds potential for real reduction in impacts to mangroves. Again, considering the most important drivers affecting mangroves, aquaculture, urbanization, and coastal development, these are factors that are amenable to policy intervention at various levels of governance. Indeed, Innis et al. (2016) recognizes that legislation is a viable avenue for protection of mangroves and cites examples of where this has been implemented. However, as these drivers also associated recognized socio-economic benefits, complete elimination of impacts is unlikely.

Innis et al. (2016) suggest a number of avenues for potential mangrove conservation, including: legislation; conventions and protected areas; management, education and restoration projects; and emerging conservation strategies, such as Reducing emissions from deforestation and forest degradation plus. These are all approaches that could be incorporated into policy developments under the Policy Reform scenario. These are also approaches that could be instituted at a variety of governance levels and are more amenable to including NCP that are not as easily monetized, such as preservation of human life from severe storms. The approach described by Datta et al. (2012) is a clear example of using a decided paradigm shift, including local stakeholder input that resembles the Policy Reform scenario. Under such an approach, a balancing of social, economic, and cultural interests would be possible and could optimize the NCP of mangroves.

Seagrasses. Seagrasses are the only flowering plants (class Monocotyledoneae) that are found in the marine environment. They are present in all continents except Antarctica (Green & Short, 2003). In spite of the low global species diversity of seagrasses (72 species of seagrasses distributed into six families, (Short et al. 2011)) compared with the terrestrial angiosperms (250,000 species approx.), these marine flowering plants can have distributional ranges that extend for thousands of kilometers of coastline along 6 geographical bioregions: 1) Temperate North Atlantic, 2) Tropical Atlantic, 3) Mediterranean, 4) Temperate North Pacific, 5) Tropical Indo-Pacific, and 6) Temperate Southern Oceans (Short et al., 2007). These widespread marine angiosperm evolved from terrestrial origins and have been present in the marine coastal waters for over 100 million years (Les et al., 1997); they constitute one of the richest and most important coastal habitats (Short et al., 2011), ranked among the most valuable ecosystems on Earth (Costanza et al., 1997, 2014).

Seagrass beds provide key ecological functions for maintaining healthy estuarine and coastal ecosystems (Cullen-Unsworth & Unsworth, 2013; Duarte et al., 2008; Moore & Short, 2006), enhancing biodiversity and water quality in the immediate environment and adjacent habitats (Duarte, 2002; Green & Short, 2003; Beaumont et al., 2007). Their canopies enhance the settlement of suspended particles and prevent resuspension; their root systems help to bind sediments over a long-term; and they release oxygen from photosynthesis. Their above and below ground systems also have a major role in coastal protection; holding and binding sediments, they prevent the scouring action of waves directly on the benthos, thus seagrasses, likewise mangroves and corals, dampen the effects of wave and current energy, reducing the processes of erosion and turbidity and increasing sedimentation (Green & Short, 2003).
Seagrass meadows, corals and mangroves, supply habitat, shelter and breeding ground for important marine species, including numerous commercially important fish and shellfish species (Hughes et al., 2009; Orth et al., 2006). In addition to these nursery functions, seagrass beds are also feeding ground for protected species (Christianen et al., 2013) and seabirds (Shaughnessy et al., 2012). Thus, seagrasses and mangroves and corals, contribute to various trophic levels of the soft-sediment coastal ecosystems enhancing overall productivity and biodiversity (Green & Short, 2003).

Summarizing, these units of analysis provide a wide range of ecosystem services, including raw materials and food, coastal protection, erosion control, water purification, maintenance of fisheries, carbon sequestration, and tourism, education, and research (Figure 5.19). Apart from providing a wide array of ecosystem services, aquatic angiosperms are valuable biological indicators integrating environmental impacts over measurable and definable timescales (Martínez-Crego et al., 2008; Orth et al., 2006). Under a changing climate context, their regulation service on organic matter accumulation could play a critical role in long-term carbon sequestration. As perennial structures, seagrasses are one of the few marine ecosystems which store carbon for relatively long periods (Green & Short, 2003). Therefore, these coastal plant communities could play an important role in climate change mitigation and adaptation (Duarte et al., 2013), not only in carbon sequestration (Fourqurean et al., 2012) but also in coastal protection (Ondiviela et al., 2014).

However, estuarine and coastal habitats have been historically altered and degraded (Halpern et al., 2008) and seagrass beds in particular, are undergoing a global decline (Waycott et al., 2009a). Seagrasses and their NCP are subjected to many pressures, both anthropogenic and natural (Green & Short, 2003) (Figure 5.19). Natural causes of seagrass decline include geological (i.e. coastal uplift or subsidence); meteorological events (i.e. major storm events); and specific biological interactions (e.g. eelgrass wasting disease) (Muehlstein et al., 1991) (Figure 5.19). Whereas, human induced threats are now widespread (Green & Short, 2003). Without considering climate change and its consequences, anthropogenic impacts range from estuarine and coastal habitat degradation; direct impact inducing fragmentation or loss of seagrass beds; increase of nutrient and sediment runoff; introduction of invasive species; hydrological alterations; and commercial fishing practices (Orth et al., 2006) (Figure 5.19). Although seagrass declines have been related to a combination of impacts rather than individual threats (Orth et al., 2006), two major causes of loss were identified by Waycott et al., 2009: direct impacts from coastal development and dredging activities; and indirect impacts from declining water quality, i.e eutrophication (Dennison et al., 1993; Krause-Jensen et al., 2008; Short & Burdick, 1996).

Due to the above mentioned multi-drivers of change, seagrass meadows are among the most threatened ecosystems, with loss-rates comparable to those reported for mangroves, coral reefs, and tropical rainforests (Waycott et al., 2009a). Their habitat is being lost and fragmented overall (Duarte, 2002; Hughes et al., 2009); over the last two decades, up to 18% of the documented seagrass area has been lost (Boudouresque et al., 2000; Green & Short, 2003; Kirkman, 1997; Short et al., 2006), with rates of decline accelerating in recent years (Waycott et al., 2009a). This present situation of declining seagrasses may be exacerbated by increasing human induced pressures (Nicholls et al., 2007; Wong et al., 2014) and additional global change drivers (Short & Neckles, 1999), including global warming (Jordà et al., 2012a) and sea level rise (Saunders et al., 2013). Considering the key role of seagrasses in the ecosystem function, their decline might be detrimental to those species that depend on them, including economically important fishes and invertebrates (Hughes et al., 2009); and considering moreover, that seagrass meadows are often dominated by a single seagrass species, the loss of only one seagrass species might initiate a negative cascade of effects for the whole biome (Duarte, 2002; Hemminga & Duarte, 2000).
Recent climate change has already impacted marine environments with documented effects on the phenology of organisms; the range and distribution of species; and the composition and dynamics of communities (Richardson et al., 2012). In the coming decades, coastal systems and low-lying areas will increasingly experience adverse climate-related impacts (IPCC, 2014). Global mean upper ocean temperatures have increased over decadal times scales from 1971 to 2010, with a global average warming trend of 0.11 ºC per decade in the upper 75 m of the ocean (IPCC, 2013b). The global ocean is predicted to continue warming during the 21st century (Collins et al., 2012) and it is very likely that, by the end of the century, over 95% of the world ocean, regional sea level rise will be positive (Church et al., 2011).

Pressures to seagrasses derived from global climate change have been extensively summarized (Björk et al., 2008; Duarte, 2002; Short & Neckles, 1999). Among the overall potential impacts of climate change, three major threats are associated with intertidal habitat forming species: increases in sea surface temperature (e.g. Jordà et al., 2012), sea level (e.g. Saunders et al., 2013), and frequency and intensity of storms together with their associated surge and swells (e.g. Ondiviela et al., 2014).

Intertidal and near-shore benthic habitats are characterized by strong vertical patterns in the distribution of organisms (Harley & Paine, 2009), being elevation relative to mean sea level a critical variable for the establishment and maintenance of biotic coastal communities (Pascual & Rodriguez-Lazaro, 2006). Consequently, zonation patterns are likely to shift following the environmental changes (Lubchenco et al., 1993). Wernberg et al. (2011) found several large and common species retreated south in seaweed communities, which could have substantial negative implications for ecological function and biodiversity.

Temperature has important implications on the geographic patterns of seagrass species abundance and distribution (Walker, 1991), being considered as one of the main variables controlling the seagrasses distribution at global scale (Greve & Binzer, 2004). Waycott et al (2007) predicted that the greatest impact of climate change on seagrasses will be caused by increases in temperature, particularly in shallower habitats where seagrasses are present.

Temperature increase may also alter seagrass abundance through direct effects on flowering and seed germination (Jordà et al., 2012a; Massa et al., 2009; Olsen et al., 2012). Since changes in seawater surface temperature would differ geographically the effects would vary between locations and therefore, some meadows could be favoured by the temperature increase; e.g. Hootsmans et al. (1987) found experimentally that temperatures rising from 10 ºC to 30 ºC significantly increased *Zostera noltii* seed germination. Short and Neckles (1999) concluded that, under global climate change, an average annual temperature increase will decrease productivity and distribution of seagrass meadows growing in locations with temperatures above the optimum for growth, or near the upper limit of thermal tolerance. In this sense, projections of future distribution of the intertidal seagrass *Z. noltii* performed using a highly accurate habitat suitability model based on mean and minimum seawater surface temperature, showed that the changes in seawater surface temperature derived from global warming would promote an important change in the distribution of the species, triggering a poleward shift of 888 km in the area suitable for the species by the end of the 21st century (Valle et al., 2014).

This shift in the species' distribution would turn into a reduction of the species climatic niche: those populations under seawater surface temperature thresholds higher than the temperature ranges required by the species (i.e. southernmost populations) would become extinct by 2100, and the colonization of the predicted suitable areas in the northernmost estuaries could be unlikely because *Z. noltii* populations have shown a low recolonisation rate from estuary to estuary (Chust et al., 2013; Diekmann et al., 2005) and might not shift their suitable habitat northward at a pace comparable to warming rates, especially in regions where the species is restricted to intertidal estuarine zones. Koch et al. (2013) also stated that many seagrass species living close to their thermal limits will have to up-regulate stress-response systems to tolerate sub-lethal temperature exposures. Therefore, physiological capacity of adaptation of the species would determine the vulnerability degree of seagrasses to climate change. Although photosynthesis and growth rates of marine macro-autotrophs are likely to increase under elevated carbon dioxide, its effects on thermal acclimation are unknown (Koch et al., 2013). Jordà et al. (2012b) reported that it is unlikely that enhanced carbon dioxide may increase seagrass resistance to disturbances such as warming.

Eutrophication is a major threat to submerged aquatic vegetation, and with more people living near the coast and the high costs of controls, the likelihood is that submerged aquatic vegetation will continue a downward trend. However, in some areas that have undergone restoration and controls on nutrients, such as Chesapeake Bay in the USA there has been some recovery (http://www.chesapeakebay.net/indicators/indicator/bay_grass_abundance_baywide). In cases where nutrient limitations are implemented, recovery is a very slow process, involving the replacement of fast-growing macroalgae with slower-growing plants. Simulation models predict recovery times of several years for fast-growing seagrasses to centuries for slow-growing seagrasses following nutrient reduction (Duarte, 1995).

Scenarios archetypes for seagrassess are very similar to those for mangroves, under Fortress World and Market Forces direct and indirect pressures to seagrasses will increase and additional global change drivers will take place, thus seagrasses will continue to suffer losses. NCP of seagrasses are not as recognized as those from mangroves and therefore an extreme acceleration of impacts might occur.

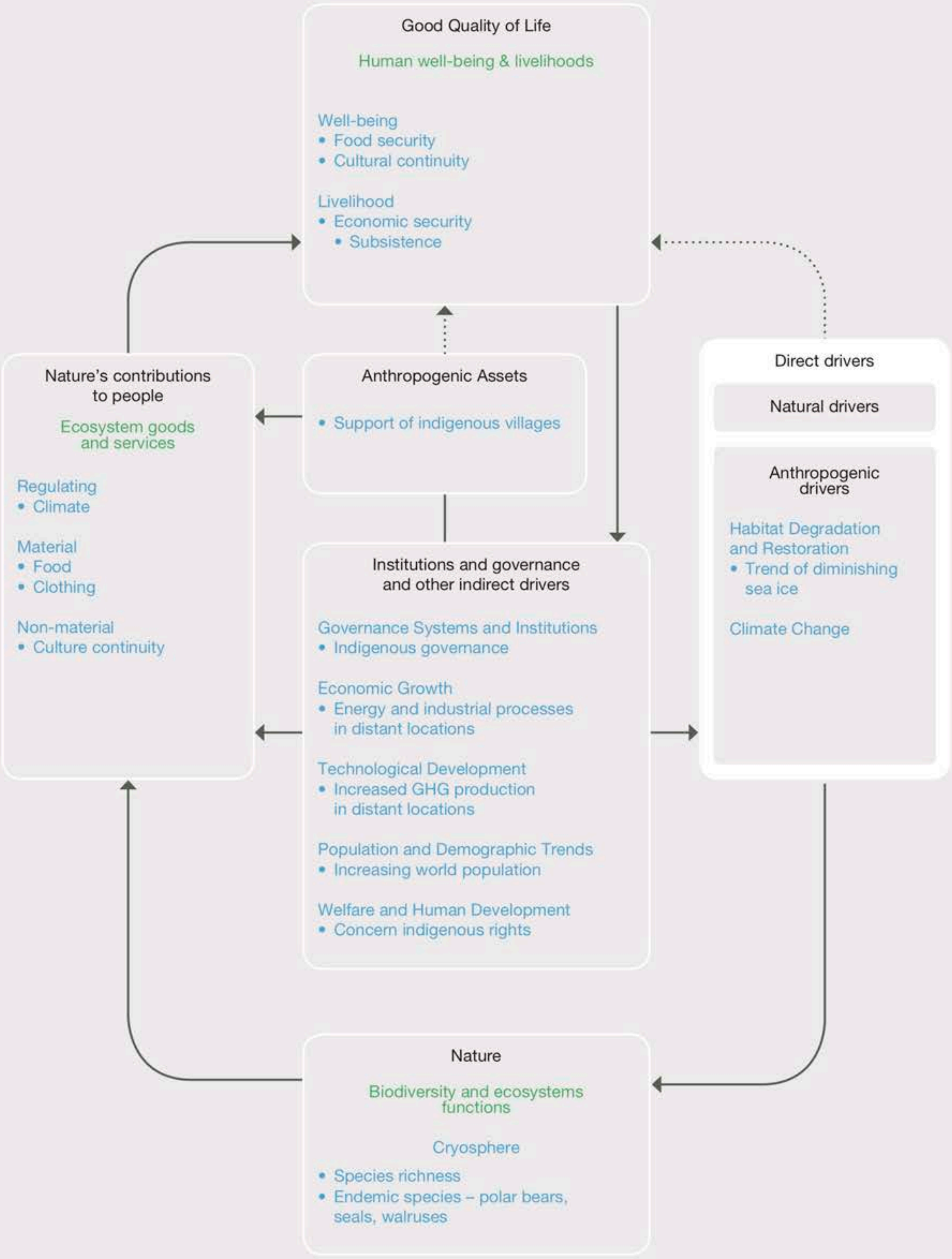
Even though present declining trends in seagrasses exceed more than 10 times the increasing trends (Waycott et al., 2009a), water quality improvements and habitat remediation are leading to encouraging results regarding the potential of seagrasses to recover (Barillé et al., 2010; Dolch et al., 2013). Thus under a Policy Reform scenario archetype, a reduction in impacts to seagrasses might be possible.

Under a Great Transitions scenario archetype where there is a high awareness and concern about the negative repercussions derived from the loss of biodiversity and ecosystem functions of these habitats, policy changes that allow seagrasses to become targeted for conservation and restoration, and promotes attenuation of global warming, seagrass meadow decline might be reduced and recovery might occur.

Salt Marshes. In recent years, many previously healthy marshes in the Americas show adverse effects from sea level rise (such as ponding, where water remains on the marsh surface during low tide and plants get water-logged), and it is questionable whether they will be able to keep up. The actual rate of sea level rise in the future will affect which marshes can keep up. Other marshes are being restored, a very expensive procedure. There are some attempts to increase their elevations (Ford et al., 1999), but given the inevitability of sea level rise at an accelerated rate, it is highly probable that extensive areas will continue to be lost. The invasive reed, *Phragmites australis*, which has reduced plant diversity in many brackish marshes in the East coast of the USA and is often removed in restoration projects, allows marshes to increase their elevation more rapidly (Rooth & Stevenson, 2000) and might better enable marshes to keep up with sea level rise.

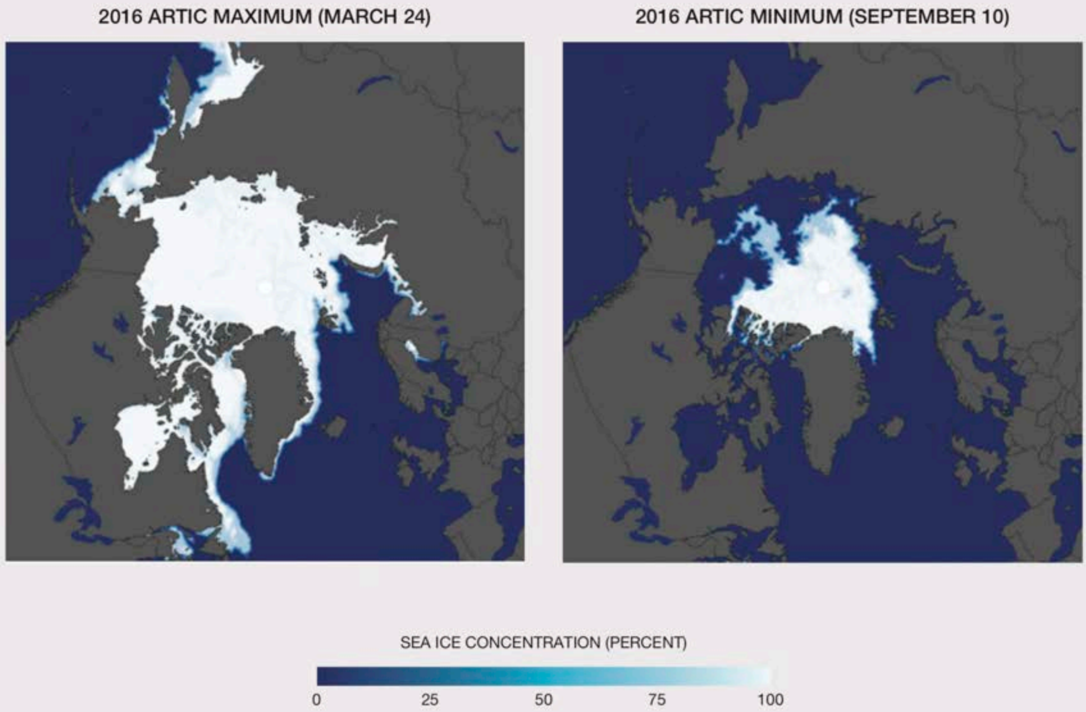
5.4.12 Cryosphere unit of analysis

Figure 5 20 Cryosphere unit of analysis viewed in the IPBES conceptual framework. Source: own representation.



Arctic sea ice is an important habitat for many species in northern Canada and Alaska. Sea ice includes both multi-year ice (fast ice) as well as seasonal ice. The extent of multi-year sea ice in the circumpolar north is highly variable and subject to cyclical drivers such as the North Atlantic Oscillation (Delworth et al., 2016). The range in area of sea ice varies on a yearly basis from 15 million km^2 on average to 7 million km^2, considering September as reference. However, this is theorized to be changing due to climate change.

Figure 5 21 Artic Sea ice conditions during the winter and summer seasons (year 2016). Source: https://earthobservatory.nasa.gov/Features/SeaIce/page3.php.



Sea ice provides provisioning services as habitat for many arctic species including polar bears, seals, and walruses, which in turn provide economic and health benefit to northern peoples. Sea ice is also considered to be providing regulating services related to climate change impacts (e.g. regional and global air and water temperatures) (Parmentier et al., 2013). There are also valuable cultural services in the form of tourism and recreation that are sometimes considered, as the arctic becomes a greater interest globally (Stewart et al., 2017).

5.5 Major trends of nature and nature's contributions to people in the Americas: learning from global scale literature

Extensive efforts have been allocated to develop global biodiversity databases and integrated assessment models with the aim of understanding past, present and future trends of nature and nature's benefit to people (e.g. Alkemade et al., 2009; Leadley et al., 2010; PBL, 2014, 2012; Pereira et al., 2010). Results from these models aim to facilitate decision makers developing policies and strategies to achieve conservation targets and sustainable uses of natural resources. They can also be used to engage the larger public in thinking about the kind of future they really want (PBL, 2012). Although most of these databases and models have a global scope, several approaches can be used to extract the most relevant information on major trends for the Americas. Here we based our approach on the results available in raw format from the Global Biodiversity model for policy support (http://www.globio.info/), a modeling framework to calculate the impact of environmental drivers on biodiversity for past, present and future. Global Biodiversity Model for policy support was developed under collaboration between Netherlands Environmental Assessment

Agency, UNEP/Global Resource Information Database - Arendal and UNEP-World Conservation Monitoring Centre.

The Global Biodiversity Model for policy support was designed to quantify past, present and future human-induced changes in terrestrial biodiversity at regional to global scales (Alkemade et al., 2009; PBL, 2016). The time frame of the period over which projections are made is 1970 – 2050. The model is built on a set of cause-effect relationships to estimate the impacts on biodiversity through time of six human-induced environmental drivers: land use, climate change, atmospheric nitrogen deposition, habitat fragmentation, disturbance by roads and disturbance through human encroachment in otherwise natural areas (PBL, 2016). The spatial information on environmental drivers used by Global Biodiversity Model for policy support is mainly derived from the Integrated Model to Assess the Global Environment 3.0 (Stehfest et al., 2014). In the Integrated Model to Assess the Global Environment -Global Biodiversity Model for policy support framework, models of socioeconomic drivers, such as climate change, land-use change and pollution, are linked with models that analyze impacts on the environment and biodiversity allowing assessment of the impact of human induced environmental drivers on biodiversity and exploring policy options in the form of intervention scenarios to reduce biodiversity loss (IPBES, 2016). Using the Integrated Model to Assess the Global Environment - Global Biodiversity Model for policy support framework, trends in biodiversity under future plausible policy scenarios have been projected, including the expected outcome in the absence of additional policies to prevent biodiversity loss (business-as-usual scenario). The results of Integrated Model to Assess the Global Environment - Global Biodiversity Model for policy support have provided information for policymakers at the international level on current biodiversity status and future trends (Alkemade et al., 2009). Specifically, model projections have been used to analyze how combinations of technological measures and changes in consumption patterns could contribute to achieve global sustainability goals by 2050 (PBL, 2012) and to inform within the fourth Global Biodiversity Outlook how sectors can contribute to the sustainable use and conservation of biodiversity (PBL, 2014).

In Global Biodiversity model for policy support, biodiversity responses are quantified as two main indicators: Natural areas and Mean Species Abundance relative to the natural state of original species. Natural areas indicator includes calculated natural areas and forestry, excluding plantations. Mean Species Abundance indicator expresses the mean abundance of original species in disturbed conditions relative to their abundance in undisturbed habitat, as an indicator of the degree to which an ecosystem is intact (PBL, 2016). The Mean Species Abundance indicator uses the species composition and abundance of the original ecosystem as a reference situation. Mean Species Abundance values have been quantified based on a synthesis (meta-analysis) of empirical species monitoring data in disturbed habitat compared to an undisturbed reference situation, reported in comparative studies derived from the literature. It covers the following taxonomic groups: mammals, birds, amphibians, reptiles, terrestrial invertebrates and vascular plants (PBL, 2016).

To project future trends of the indicators Global Biodiversity model for policy support made use of the trend scenario derived from the baseline scenario of the third Organization for Economic Cooperation and Development Environmental Outlook (OECD, 2012) as a benchmark to construct a business-as-usual future. Additionally, the model uses 3 alternative pathways that represent possible routes to achieve the sustainability targets: (1) Global technology, (2) Decentralized Solutions, and (3) Consumption Change (Table 5.2). Under the terminology used thus far in this assessment, these three pathways roughly equate to a combination the Policy Reform and Great Transitions archetypes.

Under the trend scenario (business as usual) SDG will not be achieved; the model assumes that world development continues to be characterized by a focus on economic development and globalization (Market Forces scenario archetype) and no pro-active policies to reduce the risks associated with environmental degradation are presumed (PBL, 2012). The scenario also assumes a continuing increase in the consumption of food, the production of material goods and services and the use of energy, although with a tendency towards saturation at high-income levels (Table 5.2).

The pathways represent different ways to strengthen and direct, or redirect, the technologies, preferences and incentives in society in more sustainable directions (PBL, 2012). Each alternative pathway would achieve ambitious global sustainability targets in 2050, such as limiting climate change to 2 °C, stabilizing biodiversity loss and providing full access to energy, water and food, but differ fundamentally in their approach (Table 5.2). The first pathway (Global Technology) assumes the adoption of large-scale technologically-optimal solutions to address climate change and biodiversity loss from a “top-down” approach with high level of international coordination (PBL, 2012), under this pathway the most important contribution comes from increasing agricultural productivity on highly productive lands.

The second pathway (Decentralised Solutions) relies on local and regional efforts to ensure a sustainable quality of life from a “bottom-up” managed system where small-scale and decentralized technologies are prioritized (PBL, 2012), under this pathway the major contribution is linked to avoided fragmentation, more ecological farming and reduced infrastructure expansion. The last pathway (Consumption Change) contemplates a growing awareness of sustainability issues which leads to changes in human consumption patterns and facilitates a transition towards less material- and energy-intensive activities (PBL, 2012), this implies a significant reduction in the consumption of meat and eggs as well as reduced wastage, which leads to less agricultural production and, thus, the reduction of the associated biodiversity loss.

Table 5.2. Assumptions of business as usual, global technology, decentralised solutions and consumption change scenarios for the year 2050. Sources: PBL (2012), Visconti *et al.* (2016)

<table>
<tr><th></th><th>Business as usual</th><th>Global technology</th><th>Decentralised solutions</th><th>Consumption change</th></tr>
<tr><td>Access to food</td><td>272 million people are projected to still be undernourished by 2050</td><td>Trend</td><td>Inequality in access to food due to income inequality converges to zero by 2050</td><td>Inequality in access to food due to income inequality converges to zero by 2051</td></tr>
<tr><td>Consumption</td><td>65% increase in energy consumption in the 2010–2050, 50% increase of food consumption</td><td>Trend</td><td>Trend</td><td>Meat consumption per capita levels off at twice the consumption level suggested by a supposed healthy diet (i.e., low beef, pork intake, resulting in 10 g beef, 10 g pork and 46.6 g chicken meat and eggs per person per day) (Stehfest et al., 2009; Willett, 2001)</td></tr>
<tr><td>Waste</td><td>Stable 30% of total production</td><td>Trend</td><td>Trend</td><td>Waste is reduced by 50% (15% of production)</td></tr>
<tr><td>Agricultural productivity</td><td>Yield increase by 0.06% annually (+27% by 2050)</td><td>In all regions, 30% increase in crop yields and 15% increase in livestock ‘yields’ by 2050, compared with the Trend scenario</td><td>In all regions, 20% increase in crop yields and 15% increase in livestock ‘yields’ with least possible impacts on biodiversity (Biodiversity: Mean Species Abundance in agricultural area 40% higher than in the Trend scenario)</td><td>In all regions, 15% increase in crop yields by 2050, compared with the Trend scenario</td></tr>
<tr><td>Protected areas</td><td>No further protected areas respect to 2010</td><td>17% of each of the 7 realms; Expansion allocated far from existing agriculture</td><td>17% of each of the 779 eco-regions; Expansion allocated far from existing agriculture</td><td>17% of each of the 65 realm-biomes; Expansion allocated close to existing agriculture</td></tr>
<tr><td>Greenhouse gas emissions and decabornisation rate</td><td>Greenhouse gas emissions are projected to increase by 60% and historical annual decarbonisation</td><td colspan="3">To meet the 2 °C target, atmospheric greenhouse gas concentrations are held below 450 parts per million carbon dioxide equivalents (40% to 50 % reduction) and decarbonisation rate undergo an improvement of 4.5% to 6% (3 to 4 times the historical rate).</td></tr>
</table>

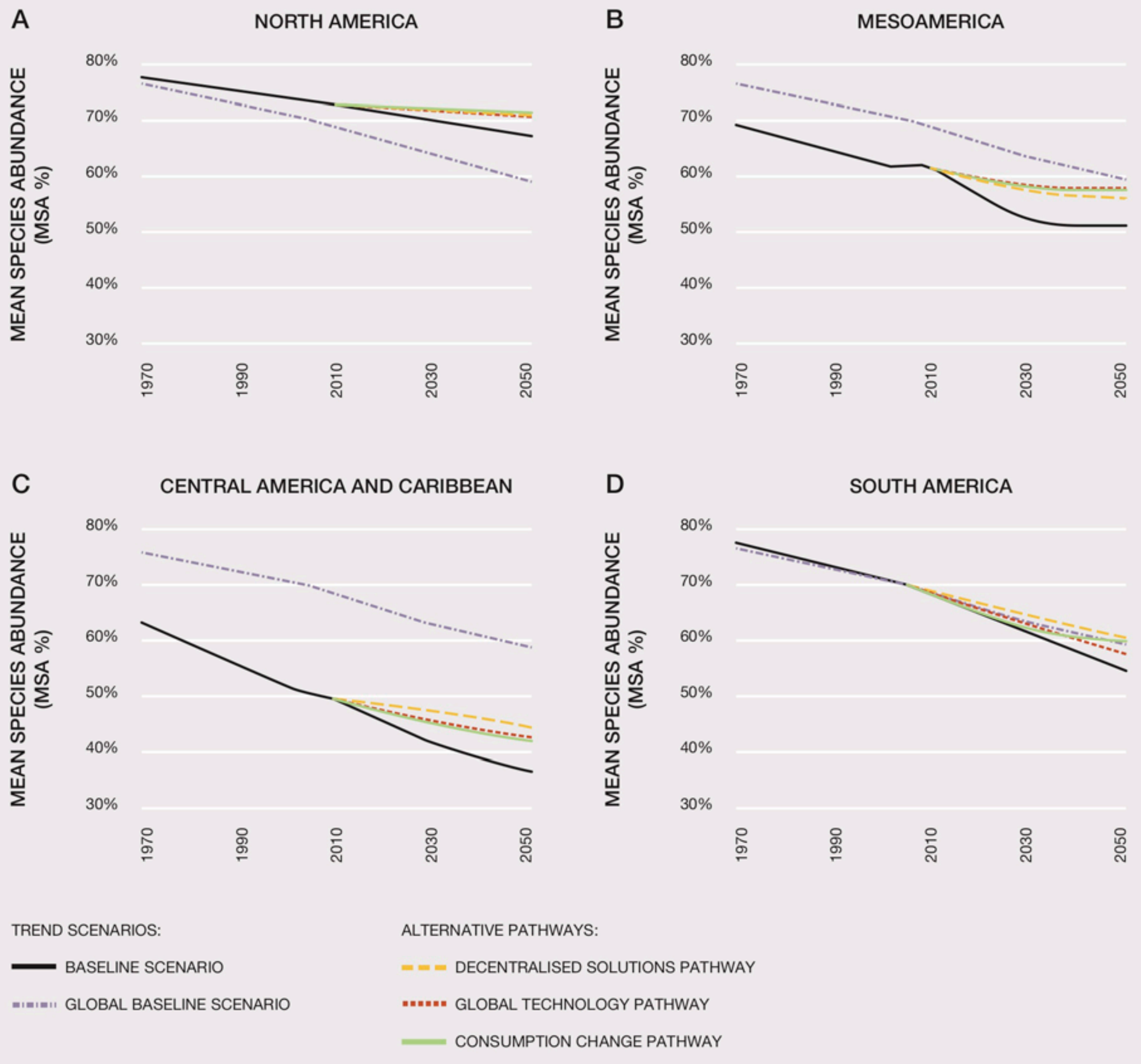
	rate of 1% to 2% is projected to continue			
Forestry	+30% in clear-cut, +35% plantation, -2.5% selective logging. No reduced impact logging	Forest plantations supply 50% of timber demand; almost all selective logging based on Reduced Impact Logging	Forest plantations supply 50% of timber demand; almost all selective logging based on Reduced Impact Logging	Forest plantations supply 50% of timber demand; almost all selective logging based on Reduced Impact Logging

Original data from Global Biodiversity Model for policy support was developed based on Integrated Model to Assess the Global Environment regions, those regions within the Americas are: (1) Canada, (2) USA; (3) Greenland; (4) Mexico; (5) Central America and Caribbean; (6) Brazil; (7) Rest of South America. In order to show a detailed picture of what is happening in the Americas region, Integrated Model to Assess the Global Environment regions have been aggregated to match as much as possible IPBES Americas subregions (North America, Mesoamerica, the Caribbean and South America) (Chapter 1). Two out of the four IPBES regions has been properly matched (1) North America, where Canada, USA and Greenland have been aggregated; and (2) South America, where Brazil and the rest of South America have been joined. The other two IPBES subregions couldn’t be represented because data cannot be disaggregated, thus data from Mexico are presented alone as a country study case, and data from Central America and Caribbean are presented together as a region.

Trends in biodiversity loss indicated by mean species abundance

Biodiversity loss, indicated by Mean Species Abundance, will continue under Trend scenarios and the three alternative pathways (Figure 5.22). Under Global Baseline scenario and Baseline scenario for the Americas, Mean Species Abundance is projected to decrease from 76% in 1970 to 59-60% in 2050. Trends in subregions from 2010 to 2050 under Baseline scenario (business as usual) show a decline from 73% to 67% for North America, from 61% to 51% for Mexico, from 64% to 37% for Central America and Caribbean, and from 68% to 55% for South America. Thus, whilst North America would experience less loss than the global and regional trends and the rest of subregions, Central America and Caribbean would experience the larger loss of biodiversity under business as usual scenario (Figure 5.22c). These declines in biodiversity could be slowed down or reduced under the 3 alternative pathways, being Desentralised Solutions the pathway leading to best results for all subregions except Mexico where Global Technology and Consumption Change could represent a better option. Under the Desentralised Solutions pathway, Central America and Caribbean could prevent their biodiversity loss by 8% compared to business as usual scenario, whereas North America and South America could reduce biodiversity loss by 5% under the same pathway and Mexico could achieve a reduction of 6% in comparison to business as usual under both Global Technology and Consumption Change pathways. In summary for the American region, under business as usual scenario, a loss of almost 40% of all original species in the Americas is expected while under the three pathways to sustainability 35 to 36% loss is presumed to occur.

Figure 5 22 **Trends in biodiversity loss indicated by mean species abundance percentage under the global baseline scenario, the trend scenario for the Americas (baseline scenario), and the alternative pathways by 2050 in A North America; B Mexico; C Central America and Caribbean; and D South America. Source: PBL Netherlands Environmental Assessment Agency (2012 and 2014).**



Trends in biodiversity indicated by natural area

Projections of biodiversity loss indicated by natural area show declining trends under Baseline scenario and the three alternative pathways, however, the projected loss by 2050 is expected to be less under the three transition pathways to sustainability in comparison to the business as usual scenario (Figure 5.23). Model projections indicate that Consumption Change pathway would lead to the best results for all regions except for the Central America and Caribbean, where Global Technology pathway could lead to a higher increase in natural area in comparison to the Desentralised Solutions and Consumption Change pathways results (Figure 5.23c). Under Consumption Change pathway Mexico could stabilise its natural areas almost to their original extent in 1970 (Figure 5.23b).

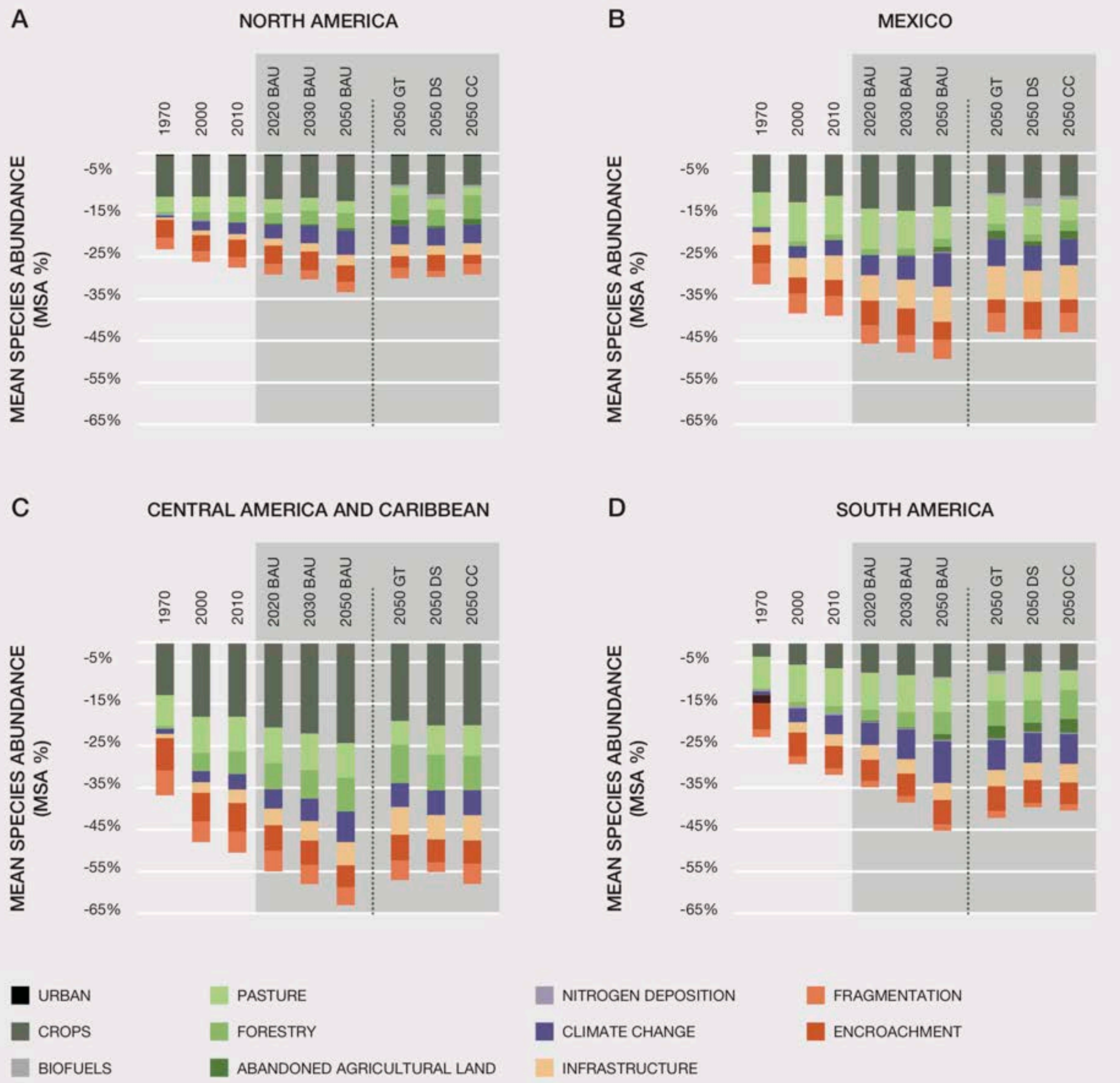
Figure 5 23 Trends in natural area percentage under the trend scenario for the Americas (baseline scenario), and the alternative pathways by 2050 in A North America; B Mexico; C Central America and Caribbean; and D South America. Source: PBL Netherlands Environmental Assessment Agency (2012 and 2014).

Pressures driving biodiversity loss

Pressures to nature are predicted to increase by 2050 under the Trend scenario (business as usual) and the three alternative pathways, negatively affecting biodiversity as indicated by a potential reduction of the Mean Species Abundance index (Figure 5.24). However, the magnitude of the pressures by 2050 is expected to be less under transition pathways to sustainability in comparison to the business as usual scenario (i.e., baseline scenario). Under the transition pathways to sustainability, climate change mitigation, the expansion of protected areas and the recovery of abandoned lands significantly contribute to reducing biodiversity loss. Although, in comparison to the projection of baseline scenario for 2050, a reduction of pressure to biodiversity driven by crops, pastures and climate change is expected under the three pathways to sustainability, other pressures to biodiversity such as forestry, biofuels and abandoned land are expected to increase. Under Baseline scenario, climate change is projected to become the fastest growing driver of biodiversity loss by 2050. The Central America and Caribbean subregion would experience larger pressures to biodiversity than the other subregions, which will be mainly driven by expansion of crops.

Figure 5 24 **Pressures driving biodiversity loss indicated by means species abundance percentage under the trend scenario from 1970 to 2050 and predicted pressures to be driving biodiversity loss under the alternative pathways by 2050 in A North America; B Mexico; C Central America and Caribbean; and D South America.**

BAU: Bussiness-as-usual; GT: Global Technology pathway; DS: Decentralised Solutions pathway; CC: Consumption Change pathway. Source: PBL Netherlands Environmental Assessment Agency (2012 and 2014).

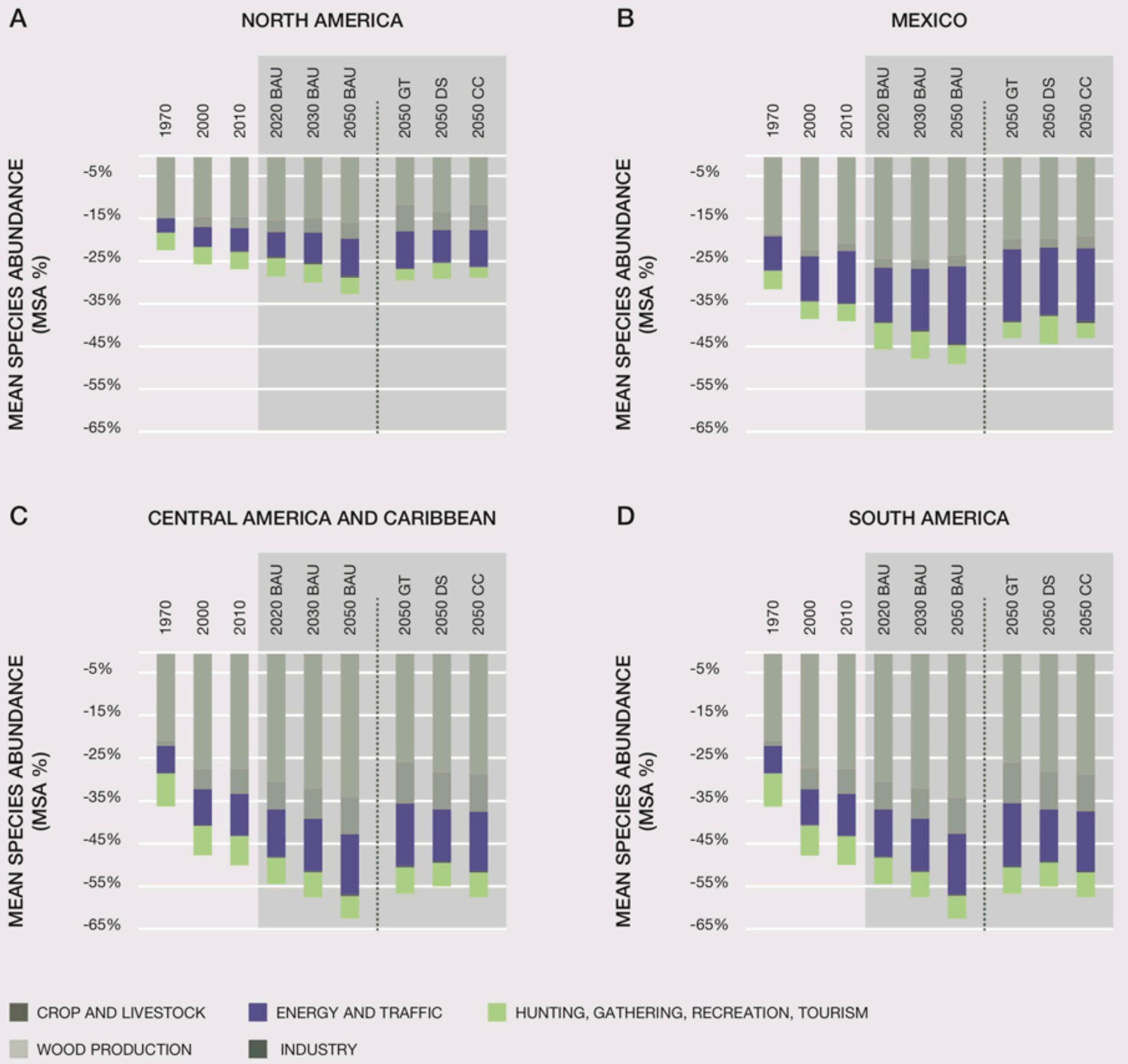


Relative share of each sector to additional biodiversity loss

Projections outputs for the baseline scenario regarding the attribution of biodiversity loss, as indicated by Mean Species Abundance percentage, to different production sectors show a similar pattern for all subregions: crop and livestock is the sector with the higher increasing trends, followed by energy and traffic sector, wood production, hunting, gathering, recreation and tourism shared sector, and industry sector (Figure 5.25). Pressures driven by those production sectors will be slowed down, or even be reduced, under the three alternative pathways, however crop and livestock will continue to have major impact in the Central America and Caribbean subregion resulting in the region with the higher percentage of biodiversity loss as indicated by Mean Species Abundance percentage.

Figure 5 25 Attribution of biodiversity loss indicated by mean species abundace percentage to different production sectors under the trend scenario from 1970 to 2050 and the alternative pathways by 2050 in A North America; B Mexico; C Central America and Caribbean; and D South America.

BAU: Bussiness-as-usual; GT: Global Technology pathway; DS: Decentralised Solutions pathway; CC: Consumption Change pathway. Source: PBL Netherlands Environmental Assessment Agency (2012 and 2014).

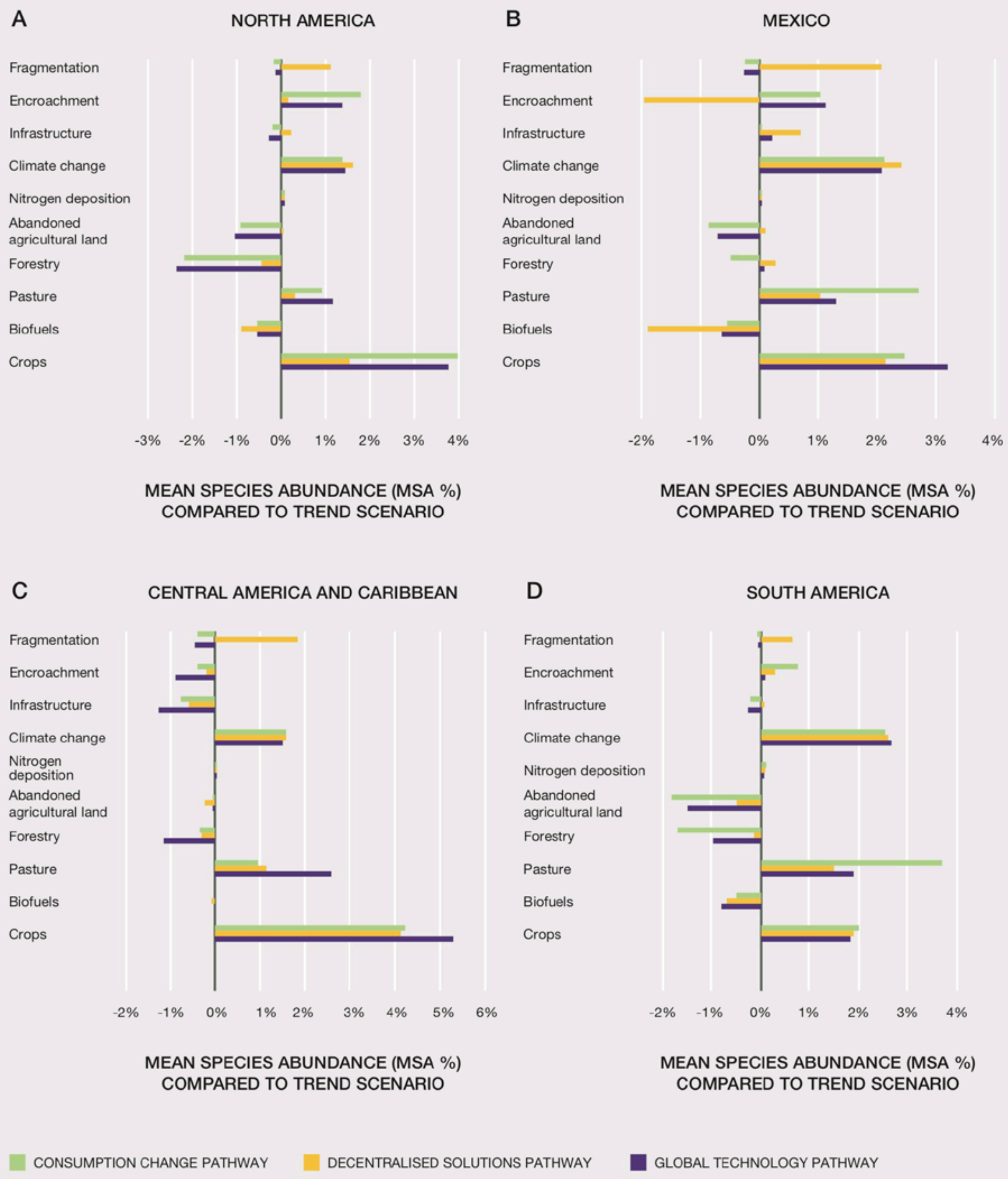


Projected relative losses of biodiversity per sector

Projected relative losses of biodiversity (Mean Species Abundance) per sector under the three different pathways, compared to trend (Baseline scenario) indicate that actions leading to land use change (reduction of crops and reduction of the use of pastures by livestock grazing) and climate change mitigation would significantly contribute to reducing biodiversity loss (Figure 5.26). As indicated above, pressures driven by forestry, demand of biofuels and abandoned land are expected to increase under the transition pathways to sustainability, which will be translated in an extra loss of biodiversity driven for those sectors in comparison to projections under business as usual scenario.

Figure 5 26 Biodiversity loss by 2050 indicated by Mean Species Abundance % compared to trend scenario in the different pathways as a consequence of changes in the different pressures: land use, climate change, nitrogen deposition, habitat fragmentation, disturbance by roads and disturbance through human encroachment in otherwise natural areas in A North America; B Mesoamerica; C Central America and Caribbean; and D South America.

GT: Global technology pathway; DS: Decentralised solutions pathway; CC: Consumption change pathway. Negative percentage values mean extra loss compared to trend and positive percentage values mean less loss compared to trend. Source: PBL Netherlands Environmental Assessment Agency (2012 and 2014).

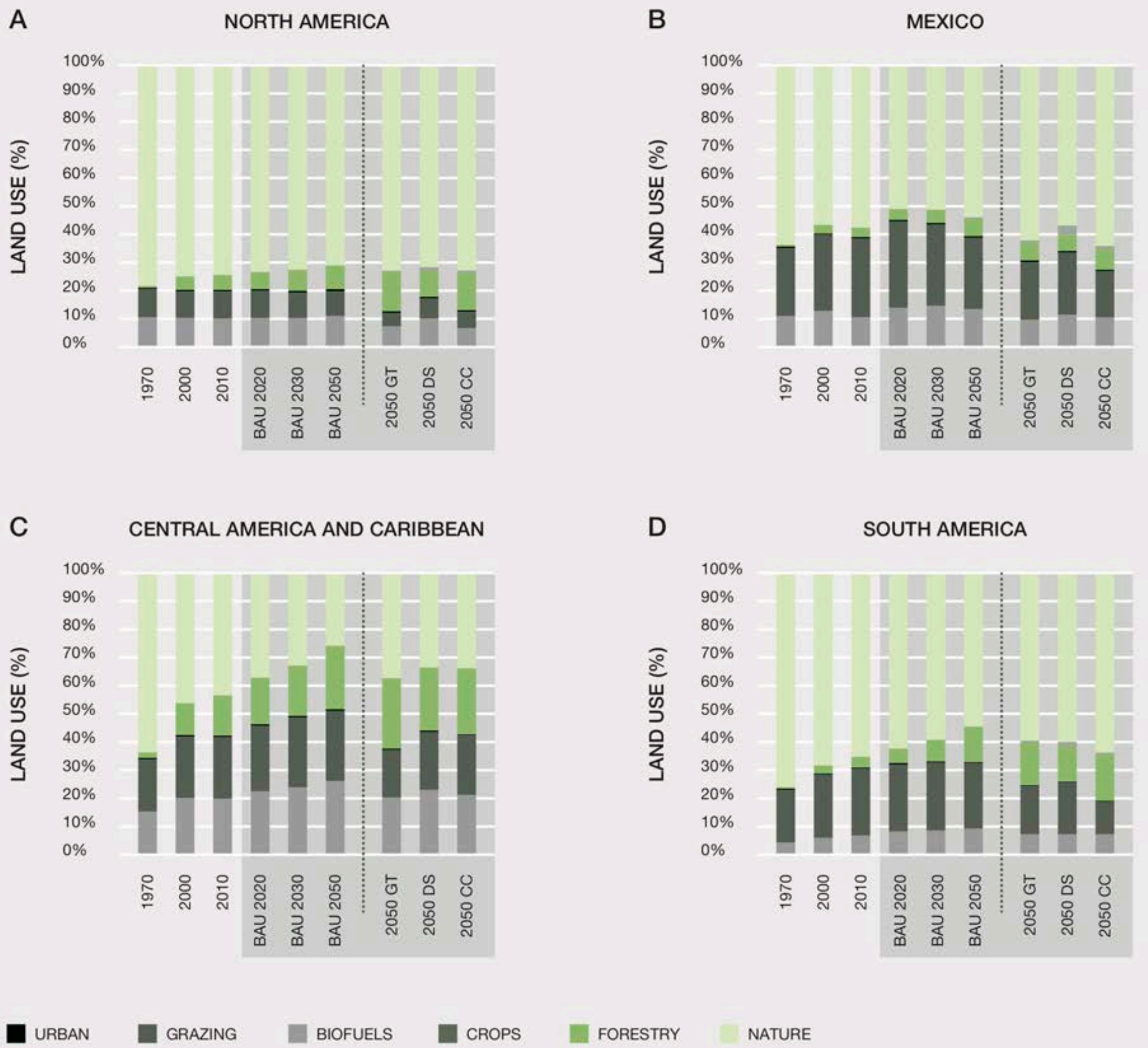


Trends in land use

According to the projected trends in land use, extent of natural areas will decrease from 2010 to 2050 under business as usual scenario in all subregions (Figure 5.27). The Central America and Caribbean subregion will experience a significant reduction in comparison to the rest. However, under transition pathways to sustainability, these trends would be reduced in all subregions by 2050. The sustainability pathways are thought to strengthen and direct, or redirect, the technologies, preferences and incentives in society to more sustainable directions, for instance to achieve the Aichi targets and the SDG. Trends in land use show that the Consumption Change pathway would lead to an increase of Natural areas within all subregions except within Central America and Caribbean, where Global Technology pathway could lead to a greater increase in natural areas. In comparison with business as usual scenario, under the Consumption Change pathways, natural area in the subregions is projected to increase 1.9% for North America; 10.1% for Mesoamerica; and 9.6% for South America, whilst Global Technology pathway would positively affect the extent of natural areas in Central America and the Caribbean by 11.2%.

Figure 5 27 **Trends in land use indicated by area percentage under the trend scenario from 1970 to 2050 and the alternative pathways by 2050 in A North America; B Mexico; C Central America and Caribbean; and D South America.**

BAU: Bussiness-as-usual; GT: Global Technology pathway; DS: Decentralised solutions pathway; CC: Consumption change pathway. Source: PBL Netherlands Environmental Assessment Agency (2012 and 2014).



In summary, according to future scenarios results presented above, it is clear that improvement of the future prospects to ensure biodiversity and NCP conservation requires rethinking the current orientation from common policies; and that change in societal options could lead to less pressure to nature and help moving towards a sustainable future. Scenarios are simplifications of complex futures, to build them several assumptions are made and these simplifying assumptions result in different limitations (Kubiszewski et al., 2017b). However, they are not intended to be predictions of the future, but rather to lay out a set of plausible futures and help decision makers and society in general, rethink possible ways to move towards more desirable futures.

5.6 Constructing a pathway to a sustainable world

Toward policy targets and Sustainable Development Goals in the face of "wicked problems"

Some problems, while not necessarily easy, are relatively straightforward, like solving an algebra problem or determining a move in chess, in which standard approaches and strategies have long been established. Then there are problems that have resisted solutions for centuries or millennia, such as human rights violations across the globe and territorial disputes. Problems of the latter category are difficult to solve primarily because their root causes are varied and complex. In social planning and management science such problems have been termed "wicked problems", not in the strict sense that they are "evil", but that they are resistant to resolution, are complicated, tend to be fraught with interdependencies, and frequently the solution to one aspect of them creates, or simply reveals, a different challenge; environmental degradation and sustainability represent wicked problems. The focal analyses of section 5.4 give a good indication of the complexity of determining what "the best" use of the world's natural capital is; multiple drivers, teleconnections, telecoupling, differing socio-economic conditions, and differences in cultures and values are all considerations in trying to create a sustainable world. The Aichi targets and the SDG represent efforts to address, or at least frame, wicked problems related to the environment and human condition, towards which solutions should be aimed.

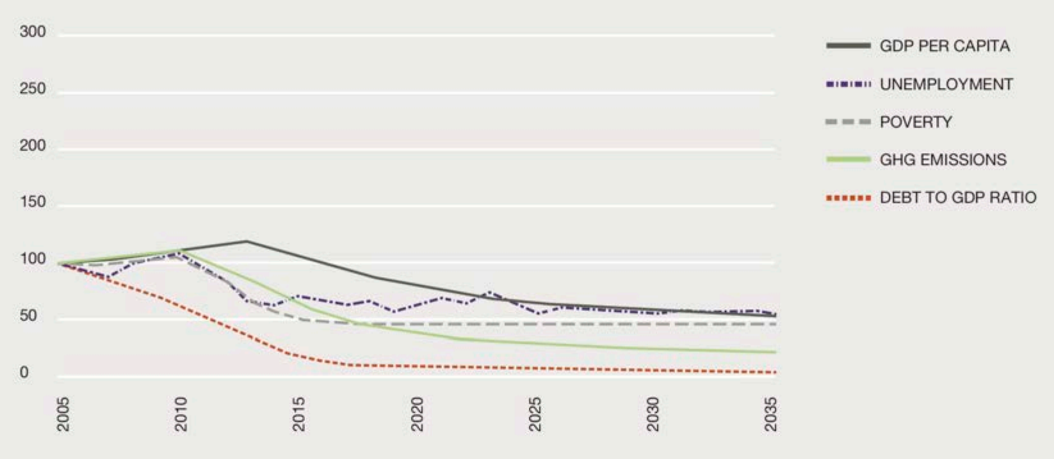
It is clear from this assessment that progress toward reaching the Aichi targets has been incremental at best and that no target has been fully reached; nevertheless they remain desirable goals. The SDG complement the Aichi targets, but the goals are still too new to expect significant progress since their promulgation. Each set of targets and goals establish guideposts on the paths to achieving the other set. Thus, solutions in one area should be designed to help provide solutions in the other. However, as both address suites of wicked problems, the question, of course, is how does policy and other decision makers actually develop solutions to meet the targets and goals? In the remainder of this section, we present a number of considerations that, based on this assessment are likely to prove helpful, if not critical, as the world goes forward in development of pathways to a sustainable future for humankind.

Box 5.2. Novel considerations and questioning the assumptions: Is it possible to achieve environmental sustainability by reducing economic growth, while increasing human well-being?

Trends show a continuing decline in biodiversity even in the most optimistic scenarios (as observed in Figure 5.22, section 5.5). Most scenarios quantifying future trends in biodiversity and ecosystem services and showing their future decline have in common a continuous growth of the economy, commonly measured as GDP (Gross Domestic Product). For example, the shared socioeconomic pathways (O'Neill et al., 2014) of the IPCC do not consider any alternative where environmental improvement goes together with low or no economic growth. In addition, evidence suggests that a decoupling of growth in both the economy (GDP) and environmental impacts is unrealistic (Ward et al., 2016). This situation has motivated a line of thought and research arguing that environmental sustainability will not be possible without considering a significant slowdown or total halt of economic growth. This has been embraced by a group of narratives that could be classified within the Great Transitions scenario archetype. For example the eco-communalism type of scenarios (Makropoulos et al., 2009), the degrowth movement (F. Schneider et al., 2010), or specific to Latin America visions like "Buen Vivir" (Gudynas, 2011). These types of visions appear as alternative pathways to development and have one main aspect in common (for a comparison see Escobar, 2015); "an equitable downscaling of production and consumption that increases human well-being and enhances ecological conditions at the local and global level, in the short and long term (F. Schneider et al., 2010)". By reducing production and consumption it is expected, indirectly, that GDP as a measure of economic growth would decline without affecting good quality of life, social equity and environmental sustainability.

There are no modeling exercises for the Americas that explicitly quantify trends into the future of NCP contemplating low economic growth. However, some modeling exercises have quantified future trends on economic and climate related aspects given energy constraints. Victor (2012) quantified a degrowth scenario for Canada in order to achieve a reduction in GDP per capita ($15,260) by 2035 (Figure 5.28). Social indicators, compared with 2005, show a reduction in unemployment and the human poverty index. Environmentally, gas emissions are reduced almost 80% by 2035.

Figure 5 28 A degrowth scenario for Canada. Source: Victor (2012).



5.6.1 Integrated scenario building

IPBES has identified scenario building as a key approach in helping decision makers assess potential future impacts of different policy options they are considering on biodiversity and NCP in an uncertain world. This is a daunting task however as no human can provide a certain prediction of what lies ahead or anticipate how existing socio-economic and environmental trends will continue, or shift unexpectedly, and the implications for vulnerable ecosystems and people. Further complications are the associated inter-linkages to consider about what all of this may mean for individual countries, subregions, regions and at the global level. Hopefully, these considerations influence the decisions that societies take in shaping the future they want. Individuals though, appear to be more interested in how decisions affect them locally. A challenge in building scenarios in the Americas is therefore to develop scenarios that have local relevance to decision makers, and that make sense in the short term demanded by political considerations, and in the long-term context required to conserve regional biodiversity and NCP.

These considerations imply that the IPBES scenarios (IPBES, 2016), should not only be built from the ground-up to the regional level, but also simultaneously from the top-down global level to the regional level. The scenarios for the Americas could therefore be conceived as being primarily focused on issues at the regional level (section 5.3) with multi-scale links down to the local level (section 5.6.3) and links up to the global level (Rosa et al., 2017) (sections 5.4 and 5.5). Although many scenario exercises have been done over time, several authors have noted that the diversity of scenarios commonly falls within predictable archetypes as outline in section 5.2. With the constraints of limited time and resources, three suggestions of how regional scenarios can be linked to typical global archetypes, such as those used by IPCC (adapted from Kok et al., 2016) are shown in Table 5.3.

Table 5.3. Strengths and weaknesses of the 3 options to develop new scenarios for biodiversity and ecosystem services as proposed by Kok *et al.* (2016).

	Strengths	Weakness
Option 1 Use existing IPCC related shared socioeconomic pathways/ RCP archetype scenarios	• readily available global pathways • can be extended to biodiversity and ecosystem services • accepted by scientists and policy makers	• minimal involvement of stakeholders • lack of connection to ILK • only implicit connection to biodiversity and ecosystem services
Option 2 Develop new global biodiversity and ecosystem scenarios	• IPBES product and opportunity to involve IPBES stakeholders • Strongly linked to biodiversity and ecosystem services • Build on results & methods of Millennium Ecosystem Assessment	• Not available yet • Requires long process with high demand for time and funding • Risk of reinventing the wheel • Difficulty of incorporating cross-scale feedbacks
Option 3 Link bottom up local biodiversity scenarios to existing shared socioeconomic pathways	• Link IPBES to existing scenarios • Explicitly multi-scale, accounting for local variability and local issues • Relatively easy to develop and connect to IPBES stakeholders	• Potential lack of cross-scale consistency and comparability • Risk of focus on local, short-term issues that could be difficult to upscale

Kok et al. (2016) further recommend Option 3 for IPBES because it builds on existing global scenarios while accommodating the heterogeneous diversity of local and regional biodiversity and ecosystem services scenarios. This proposed multi-scale scenario approach should capture the diversity of local social-ecological processes and cross-scale global-to-local interactions that affect human well-being.

While there are clear advantages of building on existing scenario work, it should not preclude new or novel approaches as they arise. As pointed out in Box 5.2, all of the shared socioeconomic pathways assume global GDP to be at current or increasing levels. However, some researchers have questioned this basic assumption (see Box 5.2).

5.6.2 Inclusion of essential stakeholder groups

Scenarios are excellent thought-provoking exercises and can help to frame pathways to a sustainable world. However, to develop plausible scenarios, and ultimately to effectuate them, like those that will be required to achieve the Great Transition endpoint, will require solutions to multiple wicked problems through the concerted efforts of at least four categories of stakeholders operating at the global/regional and local levels: 1) policy makers; 2) local populations; 3) civil society; and 4) business community. The development of plausible scenarios that can successfully drive effective policy needs to take into consideration a great many factors. As outlined in the previous section, scenario development should proceed from a regional setting with region-to-global and regional-to-local integration, which includes participation by all four categories of stakeholders. Implementation of the policies that will be necessary to fulfill a vision of a future, in which the NCP stemming from the globe's natural capital are enjoyed by all, requires the buy-in by all four categories of stakeholder; all groups are necessary to assure the plausibility of any given scenario.

Civil society may fulfill various roles in scenario development, including provision of technical expertise through scientific and academic institutions; "grass roots" organizations (formal or informal collective groups centered on an issue), conservation organizations (e.g. The Nature Conservancy, The International Union for Conservation of Nature), the IPBES effort being an example itself. So, too, civil society may play an important role representing segments of the population; i.e. providing input, or even advocacy, for particular viewpoints and may be critical in assuring consideration of particular SDG such as: 3 – Healthcare; 4 – Education; and 5 – Gender equity.

Regarding SDG, it should be recognized that SDG 4 – Jobs and economic growth, and 5 – Industrialization, are directly linked and dependent on the business community. In addition, SDG 2 – Agriculture, 7 – Energy, 12 – Production patterns, 13 – Climate change, 14 – Sustainable use of oceans and marine resources, and 15 - Forest management, all involve the business community. We speak of "natural capital" for a good reason. Aside from a pure subsistence level, realization of NCP requires higher levels of activity such as municipal and regional governments, the local business community, and multi-national corporations. The business community, at all levels, has a very decided stake in perpetuation of our natural capital and it can play a significant role in its preservation. The primary goal of most corporations is to benefit its stockholders; scenarios that do not account for this are not plausible. Thus, bottom-up scenario building needs to include not only the lowest levels of organization (i.e. the individual), but also the higher levels, such as multinational corporations. There is a significant number of forward-looking corporations that take their environmental and social responsibilities seriously and dedicate resources to those efforts. Like-minded business leaders have banded together to form such organizations as the World Business Council on Sustainability, which is composed of high-level executives dedicated to environmentally sustainable business practices. Groups such as this hold great potential in furthering the efforts of the IPBES. So too, scenario building should incorporate developing business practices such as social and environmental accounting and reporting. Other emerging trends, such as formation of "B Corporations" which have a specific recognition of social responsibility and that maximization of returns to shareholders is not necessarily their primary goal; this deviates from a principle that has been operating for over a hundred years and could produce revolutionary results in transforming the business world. Thus, in scenario building, the business world should be viewed as a resource and a necessary partner. Incorporating the views of the business community, along with other sectors of society such as local and indigenous people, will allow considering the multiple and sometimes conflicting values that often determine the effectiveness, equity and legitimacy of management and policy actions.

5.6.3 Telecoupling - Recognizing interactions between distant socio-ecological systems profoundly affect nature and nature's contribution to people

In today's highly interconnected world, sustainability issues should be analyzed with attention to the impacts that consumption and production patterns in one part of the world can have on nature, NCP and quality of life elsewhere. To do this, several concepts and frameworks have been developed with the aim of better understanding and integrating the various distant interactions that often strongly influence the flow of NCP within and between social-ecological systems, e.g. trade and invasive species. Among these, the concept of telecoupling is useful to analyze cross-scale socio-economic and environmental interactions that influence local to regional sustainability trends and outcomes (Liu et al., 2013).

Telecoupling refers to socio-economic and environmental interactions among social-ecological systems over distances and scales. The telecoupling framework takes a multilevel analytic approach. At the level of the telecoupled system, an interrelated set of social-ecological systems (sending, receiving and spillover systems) connect through flows among them. At the coupled-system level, each system consists of three interrelated components: agents, causes and effects. At the component level, each component includes many elements or dimensions, e.g. individuals, households, organizations, etc. The sustainable and equitable flow of nature contributions to people is strongly influenced by telecouplings in several socio-ecological systems of the Americas. Therefore, neglecting telecouplings and the resulting off-stage ecosystem burdens in model and scenario building, and in environmental decision-making, will jeopardize achieving SDG (Pascual et al., 2017).

Nature in many rural landscapes of Latin America has been heavily transformed in order to produce raw materials that are exported to supply the increasing demand in emerging and developed countries. Conversely, rates of environmental degradation have been reduced in some developed countries as they displace land-use abroad by importing raw materials from developing countries (Meyfroidt et al., 2013). The lower levels of environmental degradation for North America projected by the Global Biodiversity Model for policy support scenarios may be explained by the fact that the USA and Canada are large importers of food, have a large ecological footprint and thus export environmental degradation to food exporting regions such as Latin America (Moran & Kanemoto, 2017). Such telecoupling between exporting and importing regions of agricultural products means that trading decisions and policies in importing countries have a strong impact on the status of nature and its contributions to good quality of life in exporting countries.

Telecouplings can have negative or positive effects on sending and receiving systems. Many policy interventions proposed to improve sustainability outcomes in particular places (e.g. payments for ecosystem services, protected areas creation, etc.) are prone to have unintended effects on distant places, indicating that telecouplings must not be overlooked in the knowledge-policy interface (Pascual et al., 2017). Next, the telecoupling framework will be used to illustrate how cause-effect interactions between distant places influence trends and outcomes of key sustainability issues in the Americas.

Case 1: Agricultural pest control

While it is difficult to estimate true losses, reduction in agricultural crop production due to insect feeding damage ranges from 10-20% and accounts for tens of billions of USA dollars in lost harvest worldwide on an annual basis (Maine & Boyles, 2015; Oerke, 2006; Oliveira et al., 2014). It has been demonstrated that predators feeding on agricultural pests reduce feeding damage, resulting in increased yields. One such group of predators are migratory insectivorous bats.

Brazilian free-tailed bats (*Tadarida brasiliensis*) overwinter in central and southern Mexico, moving in the spring to northern Mexico and the southwestern USA, where they form large maternity colonies (aggregations of primarily female bats raising their young) and can number in the millions. They feed on a number of Lepidtopteran species (butterflies and mothes) in the family Noctuidae, including: fall armyworm (*Spodoptera frugiperda*), cabbage looper (*Trichoplusia ni*), tobacco budworm (*Heliothis virescens*), and corn earworm/cotton bollworm (*Helicoverpa zea*) (Cleveland et al., 2006). Studying the role of Brazilian free-tailed bats in a multi-county region of Texas, Cleveland et al. (2006) estimated that bats consuming 1.5 adult cotton bollworm moths per night will prevent about five moth larvae from damaging crop plants. Given that a single moth larva can destroy two to three bolls in its lifetime, they estimated that the bats reduce insect damage on cotton by 2-29%, depending on conditions.

Federico et al. (2008), in a follow-on study, calculated that Brazilian free-tailed bats not only contribute to more profitable agriculture by increasing yields, but also lower pesticide costs to farmers by delaying the build-up of cotton bollworms to critical levels, at which point pesticide applications become economical in terms of yield. Additionally, the modeling by Federico et al. (2008) indicates that predation by Brazilian free-tailed bats result in significant economic benefits even in the case of genetically modified cotton that is resistant to the moths; this has the ancillary contribution to society of lowering the amount of pesticides used.

Similar benefits from migratory, insectivorous bats for the corn crop have been shown in the Midwest of the USA. In areas where the eastern red bat (*Lasiurus borealis)*, believed to be the primary species of bat feeding on pests, was excluded from cornfields, Maine and Boyles (2015) found a 59% increase in the number of larvae of corn earworms. They calculate that for corn alone, bats reduce crop loss by over $10 billion per year worldwide. As with the Brazilian free-tailed bats, eastern red bats are migratory, overwintering in the southern USA and traveling northward to the Midwest in the spring.

There are several important points to note about these cases of telecoupling: 1) the bats spend a large portion of the year distant from where they provide benefit; 2) the beneficiaries of the cotton crop are, in essence, distributed worldwide; and; 3) being migratory, the bats are at risk not only in their summer habitat, but also during migration and in their winter habitat.

The risk to migratory bats can be substantial. Bat fatalities at wind turbines in North America have been documented at various rates, depending on the site and situation, with higher rates being reported in the Eastern USA (National Academy of Science, 2007). Strickland et al. (2011) reviewed fatality rates and found them to vary from 0.07-39.7 fatalities/MW/Year, with the highest rates associated with forested, mountain ridge tops. (Frick et al., 2017) has estimated that deaths due to wind turbines pose an actual extinction threat for some species. Fatalities can result from either direct interaction with wind turbines, i.e. bats struck by turbine blades or colliding with monopoles (Kunz et al., 2007), or from barotrauma, i.e. lung damage resulting from rapid decompression due to turbulence associated with wind turbines (Gorell et al., 2004). Approximately 75% of bat mortality associated with wind turbines in North America is accounted for by three species: eastern red bat, hoary bat (*Lasiurus cinereus*), and the silver-haired bat (*Lasionycteris noctivagans*), all of which are long-distance migrators, wintering in the southern USA and migrating north to the Midwest each summer (National Academy of Sciences Agencies, 2007). Klatt and Gehring (2013) have shown that in an agricultural area in southern Michigan USA, these three species tended to be found over open agricultural fields as opposed to riparian areas, which are preferred by the cave-hibernating bats in the area. In the Midwest, most wind farms are located within agricultural fields. Thus, preservation of NCP in agro-ecosystems can be aided by conservation of migratory, insectivorous bat species, but, ironically, these species are threatened by alternative energy options.

Case 2: Amazon forest as provider of global services

The case of the Amazon forest may well illustrate cross-scale interaction where decisions on land use at the local level may influence the global wellbeing. There have been two (intertwined) ways to look at how this influence happens: by understanding the loss of a given ecosystem service (e.g. negative consequences of deforestation for biodiversity and ecosystem functioning, or, as put by Costanza et al. (1997) and Fearnside (2008), what it would cost to replicate the service in a technologically produced, artificial biosphere, or by assessing the value of a given ecosystem service to society (e.g. the willingness to pay for an ecosystem service). In any case, different time scale analysis plays an important role for decision-making. For example, land use change from forest to pasture could show advantages in the present time (and at the local scale) (Foley et al., 2005); but be proven otherwise in the long run with implication ranging from local to regional or even global scales.

As the world's largest tropical forest (~5.4 million km^2), Amazonian forests, a myriad of biodiversity, have a substantial influence on regional and global climates (Malhi et al., 2008; Ometto et al., 2011; Schwartzman et al., 2012). For instance, almost 1/3 of the global net primary productivity (photosynthesis minus plant respiration) interannual variation is associated with Amazonia carbon fluxes (Zhao & Running, 2010). The carbon stock, in living biomass, is considered to be on the order of 150–200 Pg C, being one of the largest ecosystem carbon pool (Brienen et al., 2015; Feldpausch et al., 2012; Nogueira et al., 2015). The range of carbon pool estimate (Malhi et al., 2009; Potter et al., 2009; Saatchi et al., 2007), as well as the differences representing the vegetation cover (Bustamante et al., 2016; Ometto et al., 2014), reflects the difficulty to estimate forest structure and vegetation biomass, in a large and highly diverse ecosystem.

The carbon budget and the regional hydrological dynamic are affected by direct anthropogenic actions, as land cover and land use changes (e.g. deforestation, forest fires, forest degradation associated to unplanned logging, expansion of pasturelands) and by climate-induced extreme events, such as extended droughts (Marengo et al., 2004). Effects of these, independently or combined, increase the risk of disruption of these natural processes, as well the threat to biodiversity and ecosystem services (Aragão et al., 2014; Poulter et al., 2011). Climate feedback of these processes have also been shown through local observation and modelled at regional scale (Marengo et al., 2004; Spracklen & Garcia-Carreras, 2015), as a strong indication of the importance of the natural vegetation as climate regulation. Therefore, deforestation can, itself, be a driver of climate change (Cardoso et al., 2009; Malhi et al., 2008; Sampaio et al., 2007) at both local and global scale (Lawrence & Vandecar, 2014; Maeda et al., 2015; Werth, 2002). Normally, climate change simulations consider deforestation in large areas, or even at biome scale, although, the effect on loss of ecosystem services at local scale can drive deep changes in subregion climate, possibly weakening the resilience of the whole region (Malhi et al., 2008).
Despite the recent reduction in deforestation rates in the Brazilian Amazon, deforestation and forest degradation are still process of high concern; the region has lost about 19% of its natural cover and has about 40% of its area on conservation units and Indian reservation (Aguiar et al., 2016). The Amazon monitoring systems of Brazilian Government, as Amazon Forest Degradation Monitoring System (INPE, 2014, www.inpe.br) and Amazon Deforestation Monitoring System (INPE, 2017) identified, in the period from 2007 to 2013, illegal logging and anthropogenic fire activities, degraded 103,000 km^2 of forests, whilst clear cut deforestation impacted 56,000 km^2. From the clear cut, about 60% turned into pasturelands, and 23% is abandoned, leading to the recovery of secondary vegetation (TerraClass, INPE, 2015, www.inpe.br). These systems, associated with the agricultural census, provided useful information on the major characteristic of the rural properties, which reflected in a better mapping of the deforestation paths and characteristics (Godar et al., 2015). Those initiatives were associated to a Government act named (in Portuguese), "*Plano de Ação para Prevenção e Controle do Desmatamento na Amazônia Legal",* Brazilian Ministry of Environment, 2004, important to reduce the rate of deforestation observed in 2004, at 27,772 km^2, to 4571 km^2 in 2012. Since then, deforestation has an increasing trend, reaching 7,893 Km^2 in 2016 (INPE, 2017). However, the revision of the Brazilian forest code might threaten, under legal terms, forests from the biomes Amazon, Cerrado and Atlantic forest, mainly by the broad possibilities of reducing the requirement to preserve natural vegetation outside the farm boundaries and the relaxation of the rules for private farms established before 2008 (Brancalion et al., 2016; Sparovek et al., 2015). The dynamic of land cover change, implementation of agricultural production areas or, otherwise, further abandonment, defines important patterns of land use in the region, with similar patterns in other forests in Latin America (Boillat et al., 2017).

Although, not advocating the maintenance of the replacement of natural vegetation, local societal needs ought to be in consideration. A deep analysis in the policies addressing environmental conservation and the relation to societal need, or poverty alleviation, shows a dichotomy (Pinho et al., 2014), indicating the need of deeper action towards a sustainable future for the moist tropical forests. Boillat et al. (2017), on analysing land systems in Latin America, identified that the dynamic of land change process in the region tends to be persistent in the future. The identification of the high value services provided by the forest in comparison to what agriculture, or beef, production does goes back more than 20 years, as observed by Chomitz and Kumari (1998) and Fearnside (1997), however, the strong historical connections to the global market (Dalla-

Nora et al., 2014), the importance of commodities for the region's economies (Lapola et al., 2013), land tenure and governance, with lack of socio-ecological inclusive strategy might lead to a persistence of depletion of natural vegetation in the region.

Aguiar et al. (2016) used several socio-economic scenarios approach to calculate future carbon emissions for the Amazon region and conclude that unless a "forest based transition economy evolves in the region the land use and forest sector in Brazil shall have a limited capacity of mitigating other sectors emissions in the next decades". Historically, for the countries in Latin America and, especially considering areas of moist and dry forests, both, deforestation and forest degradation, are important drivers of carbon dioxide emissions to the atmosphere, contributing significantly to the country emissions profiles (as observed in the past two National Communications that Brazil has submitted to the United Nations Framework Convention on Climate Change, http://sirene.mcti.gov.br).

For these reasons, Foley (2005) argues it is appropriate (in order to make more informed decisions) to balance the trade-offs between "the societal benefits (typically the short-term realization of ecosystem goods and commercially valuable commodities) against the long-term costs of ecological degradation (associated with the functioning of the ecosystem). Adding to this is the fact that, in large, NCP descend from common goods (such as clean air and water, soil formation, climate regulation, waste treatment, aesthetic values and good health), which are generally taken for granted, as they do not pass through the money economy (Costanza et al., 1997).

Case 3: Urban Telecoupling

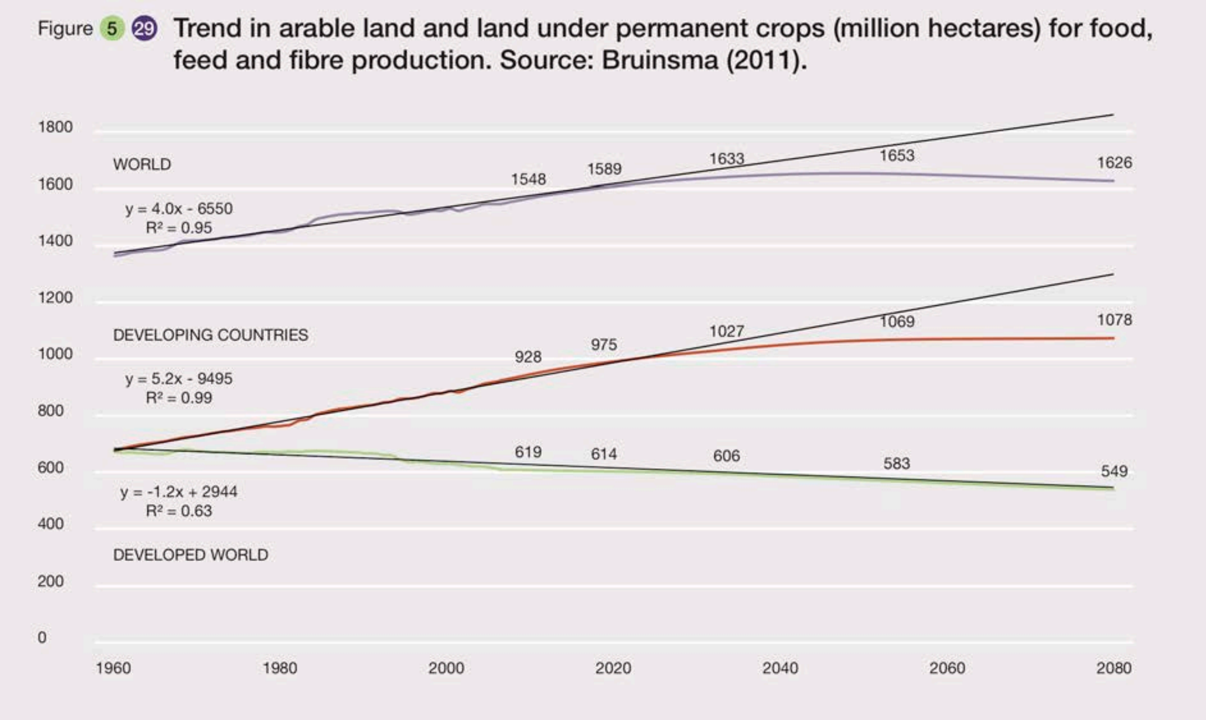
The world is increasingly urban and interconnected. This alone makes urban processes of fundamental importance to better understand global change (Huang et al., 2010) and respond to it. Today's population of 7.6 billion is expected to reach 9.8 billion in 2050, when about two-thirds of the world's population is projected to be urban (UN, 2017). This unprecedented state is posing consequences regarding the balance between demand and supply of ecosystem services in order to assure human well-being. After all, urbanization should be understood not only as a demographic or socioeconomic phenomenon but also as a process of ecological transformation by humans, affecting land ecosystems from local to global (Huang et al., 2010). This occurs for at least two intertwined reasons. First, because the increasing magnitude and pace of urbanization directly reshape land use locally in an accumulative fashion throughout the world (Seto et al., 2012). More than 1.5 million square kilometers of global urban land area is expected to be added by 2030 (Seto et al., 2011). This expansion is expected to occur at the cost of high quality agricultural land as well as high biodiversity sites (Fragkias et al., 2012). Additionally, at a global scale, the physical expansion of urban areas is growing twice as fast as urban population (Seto & Ramankutty, 2016). New expansion is expected to increasingly take place close to biodiversity hotspots. By 2030, 1.8 % additional area from biodiversity hotspots will be converted into urban use (Seto et al., 2012). It is in South America where the most pronounced increase in the amount of urban land (forecasted at 100,000 Km^2) in biodiversity hotspots will take place (Güneralp et al., 2013) and in the Americas, in general, where the highest number of species already highly threatened will be impacted by urban expansion (Seto et al., 2012).

The second reason, captured by the concept of telecoupling, is linked to trends in urban consumption patterns that unintentionally affect ecosystems at different spatial scales. However, despite conceptual advances, there is a gap in studies demonstrating these linkages. This is partially because telecoupling between places of consumption and places of production are largely unnoticed at subnational levels.

As opposed to non-urban, urban residents tend to consume differently (Gadda & Gasparatos, 2009; Rudel et al., 2009; Yu et al., 2013), artificially detached to the source of the ecosystem service. This means that urban residents, "appropriate" natural ecosystems, ecosystem goods and services, and natural capital from one or more "different elsewheres" and therefore indirectly affect land use at scales ranging from the hinterlands of the urban area to a single or multiple remote geographical unit(s) (Seitzinger et al., 2012; Seto et al., 2012). This is largely driven by economic complexities and dynamic interrelations among scales (local, regional, and global processes) and flows of goods and services. Along these lines, Seto et al. (2012) argue that since urban economies currently generate more than 90% of global gross value added, there may be few non-urban systems unaffected by urbanization. An outstanding example is the indirect impact that shifts in urban dietary preferences (Gadda & Gasparatos, 2009; Satterthwaite et al., 2010) is having on new agricultural lands and which is expected to continue growing into the future (FAO, 2017b). This is well illustrated by the growing demand for animal protein expected to continue throughout the urban world, at least until 2050. After all, more land is needed to produce meat (and dairy-based foods) than vegetable and grain-based diet (Güneralp et al., 2013). And, as demands for agricultural products grow, large remaining forest area is likely to experience increasing pressures (Defries et al., 2010; FAO, 2017b) especially in developing countries (FAO, 2017b). Therefore, not only is there a strong connection between urbanization and economic growth but also between affluence (and urban preferences) and the global displacement of land use particularly from high-income to low-income countries (Weinzettel et al., 2013). Despite increasing evidence of these trends, the underlying processes relevant to better manage the increasing telecoupled urban world are still not well captured (Liu et al., 2013; Seto & Ramankutty, 2016).

While land-use and land-cover change have been well documented, its linkage with urbanization is less well studied. As land is a finite resource, the increasing competition for land globally (e.g. for agricultural products, energy production, biomass, infrastructure and settlements, conservation and recreation, as well as a large range of other ecosystem services) and the degree of global environmental change associated with it (embedded in the general phenomenon of the “Great Acceleration”) makes the understanding between land-use and urbanization an urgent need. Most studies have focused on land-use changes driven by international food trade and its great influence on global food production and the environment. After all, agricultural products are an outstanding illustration of ecosystem services of global demand. Among studies, a particular emphasis has been around global demand for cash crops. One reported case is of continued deforestation in South America in general, and in the Amazon rainforest in particular, due to the demand for soybean (Graesser et al., 2015) by urbanized and affluent European Union countries, USA, Japan and by increasingly urbanized China, (Rudel et al., 2009; Sun et al., 2017), among others. Yu et al. (2013) show that 47% and 88% of cropland in Brazil and Argentina, respectively, are used for consumption in other countries, mainly the European Union and China. China alone displaces 5 Mha pf cropland in Brazil, mainly for soybeans. China’s appropriation of virtual water embodied in soybeans from Brazil nearly doubled between 2001 and 2007 (Liu et al., 2015). Commoditization of agriculture in the South is, therefore, a key driver affecting land cover (Lapola et al., 2013), well illustrating the interconnection and cross-scale issues of a globalized urban world. That is, telecoupling in the agriculture sector shows a very strong interaction among agri-social-ecological systems over long distances and scales.

These trends are expected to continue into the future. For example, it is expected that the demand for food between 2012 and 2050 will increase by 50%. The underlying factors will continue to be urbanization, population growth and increases in income. This increasing demand will happen as natural capacity for producing the needed food will be under increasing stress. This includes the need for additional land. It is expected that by 2050, 100 million ha of new land will be required (FAO, 2017), very likely at the expense of forested areas (e.g. natural ecosystems). This poses a threat to priority areas for biodiversity conservation in many places of Latin America, for example. In fact, the rising international demand for land embodied in food trade has been growing and is expected to continue rising throughout the coming decades, mostly at the cost of land cover conversion to new arable land in developing countries (Figure 5.29). In other words, “doubling global food supply without extensive additional environmental degradation to non urban areas presents a major challenge” (Seitzinger et al., 2012).



While cities are often solely perceived as a driver of environmental degradation, consequently affecting human well-being, they also offer important opportunities to reduce these impacts, if well managed. Therefore, urbanization has increasingly been recognized as a key element for a sustainable future, with impacts beyond urban borders. Urban environmental sustainability is now an important pillar of the new urban agenda (Habitat III, 2016). Included in the vision shared by signatories of the United Nations Conference on Housing and Sustainable Urban Development is that urban food security and strengthening of urban-rural linkages will play a major role towards sustainable urban development. Moreover, governments are committed to ensuring environmental sustainability by several measures, including the protection and improvement of ecosystem services and biodiversity.

For this end, however, the sustainability of cities needs to be understood beyond place-based concepts that advocate for decisions that are local in scope (e.g. efforts of self-sufficiency at the local level) as these decisions do not account for critical consequences of telecouplings in distant places and people. Urban telecoupling (as an analytical tool) can assist in concentrating decisions concerning urban processes (flows of capital, information, people, goods, materials, energy, and services) that spill over large geographical areas with the advantage of having both well-being and equity issues more explicit (Seto et al., 2012). Urbanization, after all, can be conceptualized as "a multidimensional, social and biophysical process driven by continuous changes across space and time in various subsystems including biophysical, built environment, and socio-institutional (e.g. economic, political, demographic, behavioral, and sociological)" (Marcotullio et al., 2014). As such, urbanization with appropriate governance, incentives, and cultural capacities (Satterthwaite et al., 2010) that adopt planetary stewardship (Seitzinger et al., 2012) may well lead the path towards a desirable global future. For example, urban residents tend to have a higher willingness to pay for ecosystem services than non-urban counterparts do. Urban citizens from Italy and the United Kingdom were willing to pay almost $44 to protect 5% of the Brazilian Amazon rain forest and therefore protect an existence value; that is, protect an ecosystem that they may not ever visit or use directly (Güneralp et al., 2013). Also, changes in urban consumption patterns can have far-reaching consequences that are less environmentally harmful. One example is the increasing European preference for organic food that has developed a new supply chain of these products in South America (Seto & Ramankutty, 2016). Moreover, urban citizens and organizations have the potential of self- organizing to ensure better decisions. The next couple of decades offer us the opportunity to showcase how cities can be responsible stewards of biodiversity and ecosystem services at all scales (Elmqvist et al., 2013).

Case 4: Biomass burn

Despite the local effect of fire, especially the high frequency of fire events in the tropical ecosystems, in general affecting biodiversity, the process of atmospheric transfer of biomass burning plume takes material and chemicals to further distances. Until 2100, atmospheric deposition of reactive nitrogen shall be the third-largest determinant of biodiversity loss, behind land use and climate changes (Sala, 2000). Plant community composition is tightly related, at larger scale, to nutrient availability, and for several ecosystems low fertility is determinant of community process stability. Therefore, changes in nitrogen input may directly impact ecosystems and constitute a major ecological threat. Among the ecological disruption processes one can highlight, nitrophilous plant species are favored in a high nitrogen input systems resulting in declining species diversity (Bobbink & Lamers, 2002); soil acidification, herbivory and susceptibly to drought, can lead to competitive exclusion and biodiversity loss.

Reactive nitrogen input in natural ecosystems, derived from atmospheric deposition is associated with several factors, such as use of fertilizer in agriculture, industrial gaseous waste/fossil fuel combustion and biomass burning. Austin et al. (2013) discuss the uneven use of nitrogen fertilizers among different countries in the Americas. In South America, especially Brazil, the use of fire is a common management practice in agricultural areas, which very often burns areas of natural vegetation marginal to the production areas. Amazonian fires contribute a flow of smoke following the jet streams associated to the Intertropical Convergence Zone, towards the southern area of the continent, including areas of Bolivia, Paraguay, Northern Argentina and substantial area of Brazil. In regions closer to highly urbanized areas, with strong industrialization, in southeastern Brazil, as well in the Central area of the Country, dominated by Cerrado biome, the nitrogen budget indicates an increase of anthropogenically derived nitrogen atmospheric deposition (Filoso et al., 2006; Lara et al., 2001).

Nitrogen deposition might affect biodiversity in priority areas for conservation in developing countries, especially in tropical and subtropical regions of the Americas. Despite the fact that the surface covered by hotspots for biodiversity conservation in these areas (2.1% of Earth's land surface), they host circa of 50% of the world's vascular plant diversity (Mittermeier et al., 2005; PHOENIX et al., 2006). Deposition rates for reactive nitrogen deposition, modeled for 2050, indicate values exceeding 15KgN ha^{-1} y^{-1} in areas of South America that are hot spots for endemic plants, as the tropical Andes and the Atlantic Forest in Brazil. Another aspect to highlight refers to the relation of nutrient availability (nitrogen and phosphorus) and carbon cycling, affecting the prediction of productivity responses of tropical ecosystems to climate changes (Cleveland & O'Connor, 2011).

Biomass burning in Southern and Eastern Brazilian Amazon, Central Brazil and Western Bolivia (www.inpe.br/queimadas) feed the atmosphere with a broad distribution of chemical compounds, including nitrogen oxides and organic substances; long-range transport of reactive nitrogen compounds are observed by smoke plume rise and transport modeling (Longo et al., 2009). This transport takes the chemical compounds to the Southern portion of Brazil, Uruguay and Northern Argentina (Zunckel et al., 2003). The photochemical reaction in the atmosphere my lead to the production of ozone, in lower altitudes, by the high nitrogen oxide presence. Ozone in lower atmosphere is phytotoxic, impacting plant communities, but also human health (Artaxo et al., 2009; Butler et al., 2008).

5.6.4 Recognition and inclusion of multiple values

Models and scenarios are powerful tools to assist in the identification of policy and management options. The arena for the design and implementation of these options is characterized by a diversity of values of nature and its contributions to people's good quality of life, associated with different cultural and institutional contexts. Stakeholders' values of nature and NCP conflict in most contexts of the Americas, affecting the way sustainability is conceived and policy and management decisions are made (Pascual et al., 2017). Thus, the full range of values should be considered when building models and scenarios if they are to assist in the development of effective, legitimate, adaptive and equitable options towards sustainability. Value conflicts arise because stakeholders hold different identities and beliefs of their relationship with nature, which produces different and sometimes contrasting preferences over NCP and ways to manage these (Mastrangelo & Laterra, 2015). Most of the literature on value conflicts tends to emphasize the dichotomy between instrumental (i.e. values of living entities as means to achieve human ends, or satisfy human preferences) vs. intrinsic (i.e. values inherent to nature, independent of human judgement) dimensions of nature (Pascual et al., 2017).

A pluralistic approach to the diversity of values underpinning nature–human relationships also recognizes that NCP can embody symbolic relationships with natural entities that define “relational values”, i.e. values that do not directly emanate from nature but are derivative of our relationships with it and our responsibilities towards it (Chan et al., 2016). Capturing this diversity of values in models and scenarios requires an integrated valuation approach. However, most valuation efforts to date have relied on unidimensional valuation approaches, by which, either economic, ecological or socio-cultural values are elicited separately. Ecological or biophysical values have been the most frequently incorporated in models and scenarios, with ecological values of multiple NCP being used in protocols for assessing and mapping NCP at regional scales such as InVEST (Nelson et al., 2009) and ECOSER (Laterra et al., 2012). Economic or monetary values have often been incorporated into models and scenarios, for example, to make global estimates of the value of ecosystems and their services (Kubiszewski et al., 2017a) (Box 5.3). In contrast, social and cultural values of nature and NCP have been rarely incorporated in models and scenarios. This represents a significant research gap as the knowledge and values of local stakeholders have been demonstrated to confer legitimacy, flexibility and adaptive capacity to policy and management actions (Pascual et al., 2017).

Integrated valuation approaches that incorporate social and cultural values allow capturing the knowledge and values of indigenous and local people. Indigenous and local knowledge can provide an important catalyst for scoping and developing management actions in response to larger-scale drivers of change (Folke et al., 2005). Given the scale of environmental problems, most efforts at building models and scenarios have been done at subregional to global scales. Incorporating ILK into these broad-scale models and scenarios becomes important as most scenario archetypes, although considering a range of drivers and impacts, make implicit assumptions on underlying worldviews and values (Kubiszewski et al., 2017a). Participatory scenario planning is one technique to incorporate multiple stakeholder values, including ILK, into models to explore plausible futures or support decisions to reach desirable futures. Participatory scenario planning is a process in which stakeholders, frequently guided by researchers, are engaged in a highly collaborative process and develop a leadership role within some or all stages of a scenario development process to investigate alternative futures (Oteros-Rozas et al., 2015). Participatory scenario planning has been applied in some socio-ecological contexts of the Americas; however, the lack of systematic monitoring and evaluation to assess its impact on the promotion of collective action and social learning precludes us from determining the actual potential of participatory scenario planning for linking broad-scale models and scenarios and ILK (Brown et al., 2016; Oteros-Rozas et al., 2015). Nevertheless, participatory scenario planning holds promise as the use of intuitive stakeholder-based scenarios rather than more formal scenarios (e.g. quantitative model outputs) reportedly engendered a greater sense of ownership of the process because participants could modify and customize narratives that incorporated local knowledge (Brown et al., 2016).

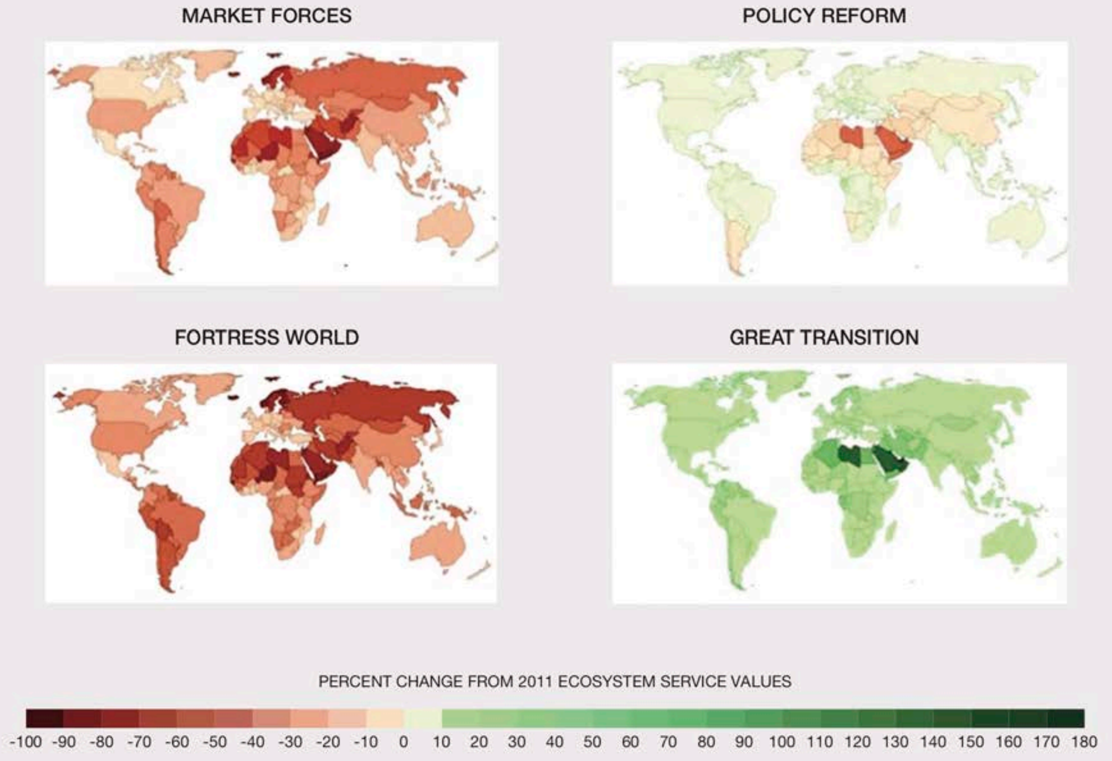
Box 5.3. Future changes in the monetary value of ecosystems services

Kubiszewski et al. (2017a) evaluated scenarios for ecosystem services in the Anthropocene globally, assessing the future change of total ecosystem services values due to land-use change decisions. The study used four scenarios archetypes of the "Great Transition Initiative" (Raskin et al., 2002) presented in section 5.2.

The change in the value of ecosystem services in each scenario was calculated considering two factors: 1) change in area covered by each ecosystem type; and 2) change in the "unit value" based on policy and management assumptions that are likely to happen in each scenario. The plausible estimates of the magnitude of change that may occur under each scenario are based roughly on the estimates from (Bateman et al., 2013) of future scenarios for the United Kingdom:

- Market Forces: 10% reduction in unit values from their 2011 levels due to a decrease in environmental and non-market factors.
- Fortress World: 20% reduction in unit values from their 2011 levels due to a significant decrease in consideration of environmental and non-market factors.
- Policy Reform: no significant change in unit values from their 2011 estimates due to a slight improvement from 2011 policies and management.
- Great Transition: 20% increase in unit values from their 2011 levels due to a significant increase in consideration of environmental and non-market factors.

Figure 5 30 **Global map showing the scale of percent change for each country in ecosystem services value in each of the four scenarios from the 2011 base map. Kubiszewski *et al.* (2017)**



Under the Market Forces and the Fortress World scenarios all countries in the Americas show a decrease in ecosystem services value (Figure 5.30), with an average negative change of 24% and 36% for Market Forces and Fortress World respectively. The highest negative percentage changes are particularly experience by islands in the Caribbean. For example, Saint Vincent and the Grenadines is expected to have a decrease in ecosystem services value of 79% under the Fortress World scenario. Within the inland countries, Bolivia shows the biggest loss (69%). In comparison, Brazil will show a decrease of 45%, equivalent to a loss of $3,717 billion/year due to losses of Tropical Forest, while USA will have a decrease of 38% ($3,279 billion/year). In the Policy Reform scenario most countries in the Americas experience an increase in ecosystem service values except for Argentina and Chile and the Caribbean islands but the magnitude of the changes are very small. In contrast, the increment in ecosystem services value is greater under the Great Transition scenario (23% average increment).

5.7 Conclusions regarding modeling, scenarios, and pathways

Scenarios and models (both qualitative and quantitative) have formed a thread throughout this chapter and we believe that several conclusions regarding their utility, use, construction, and state-of-the-art with respect to the Americas can be stated.

- While the links between the various components of the IPBES framework are easy to conceptualize qualitatively, much work remains to be done to define the relationships quantitatively, as evidenced throughout this chapter. Yet, the utility of both qualitative and quantitative modeling is clearly demonstrated by use of the IPBES framework in section 5.4 and the Global Biodiversity Model for policy support considerations presented in section 5.5, respectively.
- From Chapters 3 and 4, it is clear that region-level datasets are lacking for many taxa and drivers and this will continue to be a challenge for regional and subregional modeling in the Americas.
- Scenarios and scenario building will provide only some of the process and raw intellectual material for development of solutions for the wicked problem of biodiversity conservation. Development of new approaches to governance and new policy tools will be necessary for those solutions. Modeling will help evaluate policy options that are inherent in scenarios and both will lend themselves to development of visions of achievable and desirable futures and the most efficacious pathways to those futures. This ex-ante modeling to evaluate the effectiveness of policies is critical; as some policies and efforts may have unintended consequences.
- Scenarios are descriptions of plausible futures, but the futures themselves need to be carefully defined with clear endpoints in mind and implemented at the national and international levels. Progress is being made on defining desireable endpoints through the Aichi targets, the Paris Accord, and the SDG, but consistent with Aichi target 2, critical to the effectiveness of both is mainstreaming of the targets and goals throughout governance systems at all scales. With well-defined goals, the development of target-seekiing scenarios would likely prove productive.
- A number of considerations have been identified throughout this chapter that are necessary to insuring effective and comprehensive scenarios and modeling efforts:
    - Making use of all sources of knowledge
    - Consideration of different value systems
    - Hundreds of scenarios already exist, more effort by practitioners should go towards integration of these scenarios rather than development new ones
    - Telecoupling
    - Feedback systems in nature, especially as related to tipping points and thresholds
    - Synergies among drivers
- As with the search for modeling studies that comprehensively address the IPBES framework, no regional level visions or pathways for the Americas Region were identified through this assessment. However, a number of studies have identified principles that have met with success in more limited situations. The following are emerging principles/efforts in this area specifically from studies for the Americas.
    - Developing countries will be key factors in biodiversity conservation, as they are by definition expanding their economy, and hence, ecological footprint and have the potential

to disproportionately influence progress towards biodiversity conservation by 2050 (Adenle et al., 2014; Joshi et al., 2015).

- Participatory approaches to scenario development are helpful in insuring their achievability and the lack of participatory mechanisms can be detrimental to resource management (Bohunovsky et al., 2011; Gonzalez-Bernat & Clifton, 2017; Quinn et al., 2013; Schmitt-Olabisi et al., 2010; Seghezzo et al., 2011).
- Refocusing and directing resources in direct support of biodiversity projects, especially in developing countries, may be a viable component of future pathways (Adenle et al., 2014; Boit et al., 2016).
- Environmental management would benefit from systematic and complete reviews of available evidence and data (Cooke et al., 2016; Kremen, 2015); this concept is applicable to scenario-modeling development as well.
- Pathways, which by necessity must include socio-ecological-governance systems, can be more effective if adaptive capacity is designed into them via cooperative networks; conversely, lack of capacity can be a significant hindrance to even the best intended policies (Folke et al., 2005; Gonzalez-Bernat & Clifton, 2017; Howes et al., 2017; Joshi et al., 2015; Young et al., 2014)
- While funding plays a role in the implementation of the Convention on Biological Diversity, general awareness among policy makers also plays a significant role, whereas lack of awareness among those responsible for policy implementation can be detrimental (Gagnon-Legare & Prestre, 2014; Howes et al., 2017).

5.8 References

Abbott, B. W., Jones, J., Schuur, E., Chapin III, F. S., Bowden, W., Bret-Harte, M., Epstein, H., Flannigan, M., Harms, T., Hollingworth, T., Mack, M., McGuire, A. D., Natal, S., Rocha, A., Tank, S., Turetsky, M., Vonk, J. E., Wickland, K. P., Aiken, G. R., Alexander, H., Amon, R. M. W., & Welker, J. (2016). Biomass offsets little or none of permafrost carbon release from soils, streams, and wildfire: an expert assessment. *Environmental Research Letters*, *11*(3), 034014. https://doi.org/10.1088/1748-9326/11/3/034014

Adenle, A., Stevens, C., & Bridgewater, P. (2014). Stakeholder Visions for Biodiversity Conservation in Developing Countries. *Sustainability*, *7*(1), (pp. 271–293). http://doi.org/10.3390/su7010271

Aguiar, A. P. D., Ometto, J. P., Nobre, C., Lapola, D. M., Almeida, C., Vieira, I. C., Soares, J. V., Alvala, R., Saatchi, S., Valeriano, D., & Castilla-Rubio, J. C. (2012). Modeling the spatial and temporal heterogeneity of deforestation-driven carbon emissions: the INPE-EM framework applied to the Brazilian Amazon. *Global Change Biology*, *18*(11), 3346–3366. https://doi.org/10.1111/j.1365-2486.2012.02782.x

Aguiar, A. P. D., Vieira, I. C. G., Assis, T. O., Dalla-Nora, E. L., Toledo, P. M., Oliveira Santos-Junior, R. A., Batistella, M., Coelho, A. S., Savaget, E. K., Aragão, L. E. O. C., Nobre, C. A., & Ometto, J. P. H. (2016). Land use change emission scenarios: anticipating a forest transition process in the Brazilian Amazon. *Global Change Biology*, *22*(5), 1821–1840. https://doi.org/10.1111/gcb.13134

Aguiar, A. P., Tejada, G., Assis, T., & Dalla-Nora, E. (2014). *AMAZALERT PROJECT - Set of land-use scenarios for Brazil, linked to implications for policies: Final Report*.

Alkemade, R., Van Oorschot, M., Miles, L., Nellemann, C., Bakkenes, M., & Ten Brink, B. (2009). GLOBIO3: A framework to investigate options for reducing global terrestrial biodiversity loss. *Ecosystems*, *12*(3), (pp. 374–390). http://doi.org/10.1007/s10021-009-9229-5

Alongi, D. M. (2002). *Present state and future of the world's mangrove forests. Environmental Conservation*, *29*, (pp. 331–349). http://doi.org/10.1017/S0376892902000231

Anadón, J. D., Sala, O. E., & Maestre, F. T. (2014). Climate change will increase savannas at the expense of forests and treeless vegetation in tropical and subtropical Americas. *Journal of Ecology*, *102*(6), (pp. 1363–1373). http://doi.org/10.1111/1365-2745.12325

Anthony, E. J., & Gratiot, N. (2012a). Coastal engineering and large-scale mangrove destruction in Guyana, South America: Averting an environmental catastrophe in the making. *Ecological Engineering*, *47*, (pp. 268–273). http://doi.org/10.1016/j.ecoleng.2012.07.005

Anthony, E. J., & Gratiot, N. (2012b). Coastal engineering and large-scale mangrove destruction in Guyana, South America: Averting an environmental catastrophe in the making. *Ecological Engineering*, *47*, (pp. 268–273). http://doi.org/10.1016/j.ecoleng.2012.07.005

Aragão, L. E. O. C., Poulter, B., Barlow, J. B., Anderson, L. O., Malhi, Y., Saatchi, S., Phillips, O. L., & Gloor, E. (2014). Environmental change and the carbon balance of Amazonian forests. *Biological Reviews*, *89*(4), 913–931. https://doi.org/10.1111/brv.12088

Arkema, K. K., Guannel, G., Verutes, G., Wood, S. a., Guerry, A., Ruckelshaus, M., Kareiva, P., Lacayo, M., & Silver, J. M. (2013). Coastal habitats shield people and property from sea-level rise and storms. *Nature Climate Change*, *3*(10), 913–918. https://doi.org/10.1038/nclimate1944

Artaxo, P., Rizzo, L. V, Paixão, M., De Lucca, S., Oliveira, P. H., Lara, L. L., Wiedemann, K. T., Andreae, M. O., Holben, B., Schafer, J., Correia, A. L., & Pauliquevis, T. M. (2009). Aerosol Particles in Amazonia: Their Composition, Role in the Radiation Balance, Cloud Formation, and Nutrient Cycles. In M. Keller, M. Bustamante, J. Gash, & P. Silva Dias (Eds.), *Amazonia and Global Change* (pp. 233–250). American Geophysical Union. https://doi.org/10.1029/2008GM000847

Artic Council. (2016). *Arctic Resilience Report*. *(M. Carson & G. Peterson, Eds.).* Stockholm: Stockholm Environment Institute and Stockholm Resilience Centre.

Asner, G. P., Rudel, T. K., Aide, T. M., Defries, R., & Emerson, R. (2009). A Contemporary Assessment of Change in Humid Tropical Forests. *Conservation Biology*, *23*(6), (pp. 1386–1395). http://doi.org/10.1111/j.1523-1739.2009.01333.x

Austin, A. T., Bustamante, M. M. C., Nardoto, G. B., Mitre, S. K., Perez, T., Ometto, J. P. H. B., Ascarrunz, N. L., Forti, M. C., Longo, K., Gavito, M. E., Enrich-Prast, A., & Martinelli, L. A. (2013). Latin America's

Nitrogen Challenge. *Science*, *340*(6129), 149–149. https://doi.org/10.1126/science.1231679

Barillé, L., Robin, M., Harin, N., Bargain, A., & Launeau, P. (2010). Increase in seagrass distribution at Bourgneuf Bay (France) detected by spatial remote sensing. *Aquatic Botany*, *92*(3), (pp. 185–194). http://doi.org/10.1016/j.aquabot.2009.11.006

Bateman, I. J., Harwood, A. R., Mace, G. M., Watson, R. T., Abson, D. J., Andrews, B., Binner, A., Crowe, A., Day, B. H., Dugdale, S., Fezzi, C., Foden, J., Hadley, D., Haines-Young, R., Hulme, M., Kontoleon, A., Lovett, A. a, Munday, P., Pascual, U., Paterson, J., Perino, G., Sen, A., Siriwardena, G., van Soest, D., & Termansen, M. (2013). Bringing Ecosystem Services into Economic Decision-Making: Land Use in the United Kingdom. *Science*, *341*(6141), 45–50. https://doi.org/10.1126/science.1234379

Beach, D. M., & Clark, D. A. (2015). Scenario planning during rapid ecological change: lessons and perspectives from workshops with southwest Yukon wildlife managers. *Ecology and Society*, *20*(1), (pp. 61). http://doi.org/10.5751/ES-07379-200161

Beaumont, N., Austen, M., Atkins, J. P., Burdon, D., Degraer, S., Dentinho, T., Derous, S., Holm, P., Horton, T., van Ierland, E., Marboe, A., Starkey, D., Townsend, M., & Zarzycki, T. (2007). Identification, definition and quantification of goods and services provided by marine biodiversity: Implications for the ecosystem approach. *Marine Pollution Bulletin*, *54*(3), 253–265. https://doi.org/10.1016/j.marpolbul.2006.12.003

Bello, C., Galetti, M., Pizo, M. A., Magnago, L. F. S., Rocha, M. F., Lima, R. A. F., Peres, C. A., Ovaskainen, O., & Jordano, P. (2015). Defaunation affects carbon storage in tropical forests. *Science Advances*, *1*(11), 1–11. https://doi.org/10.1126/sciadv.1501105

Bhatt, U. S., Walker, D. A., Raynolds, M. K., Comiso, J. C., Epstein, H. E., Jia, G., Gens, R., Pinzon, J. E., Tucker, C. J., Tweedie, C. E., & Webber, P. J. (2010). Circumpolar Arctic Tundra Vegetation Change Is Linked to Sea Ice Decline. *Earth Interactions*, *14*(8), 1–20. https://doi.org/10.1175/2010EI315.1

Binding, C. E., Greenberg, T. A., Watson, S. B., Rastin, S., & Gould, J. (2015). Long term water clarity changes in North America's Great Lakes from multi-sensor satellite observations. *Limnology and Oceanography*, *60*(6), (pp. 1976–1995). http://doi.org/10.1002/lno.10146

Björk, M., Short, F. T., Mcleod, E., & Beer, S. (2008). *Managing seagrasses for resilience to climate change* (IUCN, Gland, Switzerland. 56pp.). *IUCN Resilience Science Group Working Paper Series* (Vol. 3). IUCN, Gland, Switzerland, 56 pp.

Bobbink, R., & Lamers, L. P. . . (2002). Effects of increased nitrogen deposition. In J. N. B. Bell & M. Treshow (Eds.), *Air pollution and plant life* (pp. 201-235). Chichester: John Wiley & Sons.

Bohunovsky, L., Ja, J., & Omann, I. (2011). Participatory scenario development for integrated sustainability assessment. *Regional Environmental Change*, *11*, (pp. 271–284). http://doi.org/10.1007/s10113-010-0143-3

Boillat, S., Scarpa, F. M., Robson, J. P., Gasparri, I., Aide, T. M., Aguiar, A. P. D., Anderson, L. O., Batistella, M., Fonseca, M. G., Futemma, C., Grau, H. R., Mathez-Stiefel, S.-L., Metzger, J. P., Ometto, J. P. H. B., Pedlowski, M. A., Perz, S. G., Robiglio, V., Soler, L., Vieira, I., & Brondizio, E. S. (2017). Land system science in Latin America: challenges and perspectives. *Current Opinion in Environmental Sustainability*, *26–27*, 37–46. https://doi.org/10.1016/j.cosust.2017.01.015

Boit, A., Sakschewski, B., Boysen, L., Cano-Crespo, A., Clement, J., Garcia-alaniz, N., Kok, K., Kolb, M., Langerwisch, F., Rammig, A., Sachse, R., van Eupen, M., von Bloh, W., Clara Zemp, D., & Thonicke, K. (2016). Large-scale impact of climate change vs. land-use change on future biome shifts in Latin America. *Global Change Biology*, *22*(11), 3689–3701. https://doi.org/10.1111/gcb.13355

Bonilla-Moheno, M., Redo, D. J., Aide, T. M., Clark, M. L., & Grau, H. R. (2013). Vegetation change and land tenure in Mexico: A country-wide analysis. *Land Use Policy*, *30*(1), (pp. 355–364). http://doi.org/10.1016/j.landusepol.2012.04.002

Boudouresque, C. F., Charbonel, E., Meinesz, A., Pergent, G., Pergent-Martini, C., Cadiou, G., Bertrandy, M. C., Foret, P., Ragazzi, M., & Rico-Raimondino, V. (2000). A monitoring network based on the seagrass Posidonia Oceanica in the Northwestern Mediterranean Sea. *Biologia Marina Mediterránea*, *7*, 328–331.

Bozec, Y.-M., O'Farrell, S., Bruggemann, J. H., Luckhurst, B. E., & Mumby, P. J. (2016). Tradeoffs between fisheries harvest and the resilience of coral reefs. *Proceedings of the National Academy of Sciences of the United States of America*, *113*(16), (pp. 4536–41). http://doi.org/10.1073/pnas.1601529113

Brancalion, P. H. S., Garcia, L. C., Loyola, R., Rodrigues, R. R., Pillar, V. D., & Lewinsohn, T. M. (2016). Análise crítica da Lei de Proteção da Vegetação Nativa (2012), que substituiu o antigo Código Florestal: atualizações e ações em curso. *Natureza & Conservação*, *14*(2012), e1–e16. http://doi.org/10.1016/j.ncon.2016.03.004

Brienen, R. J. W., Phillips, O. L., Feldpausch, T. R., Gloor, E., Baker, T. R., Lloyd, J., Lopez-Gonzalez, G., Monteagudo-Mendoza, A., Malhi, Y., Lewis, S. L., Vásquez Martinez, R., Alexiades, M., Álvarez Dávila, E., Alvarez-Loayza, P., Andrade, A., Aragão, L. E. O. C., Araujo-Murakami, A., Arets, E. J. M. M., Arroyo, L., Aymard C., G. A., Bánki, O. S., Baraloto, C., Barroso, J., Bonal, D., Boot, R. G. A., Camargo, J. L. C., Castilho, C. V., Chama, V., Chao, K. J., Chave, J., Comiskey, J. A., Cornejo Valverde, F., da Costa, L., de Oliveira, E. A., Di Fiore, A., Erwin, T. L., Fauset, S., Forsthofer, M., Galbraith, D. R., Grahame, E. S., Groot, N., Hérault, B., Higuchi, N., Honorio Coronado, E. N., Keeling, H., Killeen, T. J., Laurance, W. F., Laurance, S., Licona, J., Magnussen, W. E., Marimon, B. S., Marimon-Junior, B. H., Mendoza, C., Neill, D. A., Nogueira, E. M., Núñez, P., Pallqui Camacho, N. C., Parada, A., Pardo-Molina, G., Peacock, J., Peña-Claros, M., Pickavance, G. C., Pitman, N. C. A., Poorter, L., Prieto, A., Quesada, C. A., Ramírez, F., Ramírez-Angulo, H., Restrepo, Z., Roopsind, A., Rudas, A., Salomão, R. P., Schwarz, M., Silva, N., Silva-Espejo, J. E., Silveira, M., Stropp, J., Talbot, J., ter Steege, H., Teran-Aguilar, J., Terborgh, J., Thomas-Caesar, R., Toledo, M., Torello-Raventos, M., Umetsu, R. K., van der Heijden, G. M. F., van der Hout, P., Guimarães Vieira, I. C., Vieira, S. A., Vilanova, E., Vos, V. A., & Zagt, R. J. (2015). Long-term decline of the Amazon carbon sink. *Nature*, *519*(7543), 344–348. https://doi.org/10.1038/nature14283

Brooks, T. M., Akçakaya, H. R., Burgess, N. D., Butchart, S. H. M., Hilton-Taylor, C., Hoffmann, M., Juffe-Bignoli, D., Kingston, N., MacSharry, B., Parr, M., Perianin, L., Regan, E. C., Rodrigues, A. S. L., Rondinini, C., Shennan-Farpon, Y., & Young, B. E. (2016). Analysing biodiversity and conservation knowledge products to support regional environmental assessments. *Scientific Data*, *3*, 160007. https://doi.org/10.1038/sdata.2016.7

Bustamante, M. M. C., Nobre, C. A., Smeraldi, R., Aguiar, A. P. D., Barioni, L. G., Ferreira, L. G., Longo, K., May, P., Pinto, A. S., & Ometto, J. P. H. B. (2012). Estimating greenhouse gas emissions from cattle raising in Brazil. *Climatic Change*, *115*(3–4), 559–577. https://doi.org/10.1007/s10584-012-0443-3

Bustamante, M. M. C., Roitman, I., Aide, T. M., Alencar, A., Anderson, L. O., Aragão, L., Asner, G. P., Barlow, J., Berenguer, E., Chambers, J., Costa, M. H., Fanin, T., Ferreira, L. G., Ferreira, J., Keller, M., Magnusson, W. E., Morales-Barquero, L., Morton, D., Ometto, J. P. H. B., Palace, M., Peres, C. A., Silvério, D., Trumbore, S., & Vieira, I. C. G. (2016). Toward an integrated monitoring framework to assess the effects of tropical forest degradation and recovery on carbon stocks and biodiversity. *Global Change Biology*, *22*(1), 92–109. https://doi.org/10.1111/gcb.13087

Butler, T. M., Taraborrelli, D., Brühl, C., Fischer, H., Harder, H., Martinez, M., Williams, J., Lawrence, M. G., & Lelieveld, J. (2008). Improved simulation of isoprene oxidation chemistry with the ECHAM5/MESSy chemistry-climate model: lessons from the GABRIEL airborne field campaign. *Atmospheric Chemistry and Physics*, *8*(16), 4529–4546. https://doi.org/10.5194/acp-8-4529-2008

Buytaert, W., Cuesta-Camacho, F., & Tobón, C. (2011). Potential impacts of climate change on the environmental services of humid tropical alpine regions. *Global Ecology and Biogeography*. http://doi.org/10.1111/j.1466-8238.2010.00585.x

Caballero, J., Palacios, F., Arévalos, F., Rodas, O., & Yanosky, A. A. (2013). Cambio de uso de la tierra en el Gran Chaco Americano en el año 2013. *Paraquaria Natural*, *2*(1), 21–28.

CAFF. (2013). *Arctic Biodiversity Assessment - Status and Trends in Arctic Biodiversity*. *Arctic Biodiversity Assessment*.

Canada, G. of. (1991). *THE FEDERAL POLICY ON WETLAND CONSERVATION Government of Canada 1991*. Ottawa, ON.

Capellán-Pérez, I., Mediavilla, M., de Castro, C., Carpintero, Ó., & Miguel, L. J. (2015). More growth? An unfeasible option to overcome critical energy constraints and climate change. *Sustainability Science*, *10*(10), (pp. 397–411). http://doi.org/10.1007/s11625-015-0299-3

Cárcamo, P. F., Garay-Flühmann, R., Squeo, F. A., & Gaymer, C. F. (2014). Using stakeholders' perspective of ecosystem services and biodiversity features to plan a marine protected area. *Environmental Science & Policy*, *40*(40), (pp. 116–131). http://doi.org/10.1016/j.envsci.2014.03.003

Cardoso, M., Nobre, C., Sampaio, G., Hirota, M., Valeriano, D., & Câmara, G. (2009). Long-term potential for tropical-forest degradation due to deforestation and fires in the Brazilian Amazon. *Biologia*, *64*(3). http://doi.org/10.2478/s11756-009-0076-9

Cavanaugh, K. C., Kellner, J. R., Forde, A. J., Gruner, D. S., Parker, J. D., Rodriguez, W., & Feller, I. C. (2014). Poleward expansion of mangroves is a threshold response to decreased frequency of extreme cold events. *Proceedings of the National Academy of Sciences of the United States of America*. http://doi.org/10.1073/pnas.1315800111

Chan, K. M. A., Balvanera, P., Benessaiah, K., Chapman, M., Díaz, S., Gómez-Baggethun, E., Gould, R., Hannahs, N., Jax, K., Klain, S., Luck, G. W., Martín-López, B., Muraca, B., Norton, B., Ott, K., Pascual, U., Satterfield, T., Tadaki, M., Taggart, J., & Turner, N. (2016b). Why protect nature? Rethinking values and the environment. *PNAS*, *113*(6), (pp. 1462–1465). https://doi.org/doi/10.1073/pnas.1525002113

Chapin, F. S., Zavaleta, E. S., Eviner, V. T., Naylor, R. L., Vitousek, P. M., Reynolds, H. L., Hooper, D. U., Lavorel, S., Sala, O. E., Hobbie, S. E., Mack, M. C., & Díaz, S. (2000). Consequences of changing biodiversity. *Nature*, *405*(6783), 234–242. https://doi.org/10.1038/35012241

Chaplin-Kramer, R., Sharp, R. P., Mandle, L., Sim, S., Johnson, J., Butnar, I., Milà i Canals, L., Eichelberger, B. A., Ramler, I., Mueller, C., McLachlan, N., Yousefi, A., King, H., & Kareiva, P. M. (2015). Spatial patterns of agricultural expansion determine impacts on biodiversity and carbon storage. *Proceedings of the National Academy of Sciences*, *112*(24), 7402–7407. https://doi.org/10.1073/pnas.1406485112

Chapra, S. C., Dove, A., & Rockwell, D. C. (2009). Great Lakes chloride trends: Long-term mass balance and loading analysis. *Journal of Great Lakes Research*, *35*(2), (pp. 272–284). http://doi.org/10.1016/J.JGLR.2008.11.013

Chomitz, K. M., & Kumari, K. (1998). The domestic benefits of tropical forests: a critical review. *The World Bank Research Observer*, *13*(1), (pp. 13–35).

Christianen, M. J., van Belzen, J., Herman, P. M. J., van Katwijk, M. M., Lamers, L. P. M., van Leent, P. J. M., & Bouma, T. J. (2013). Low-canopy seagrass beds still provide important coastal protection services. *PloS One*, *8*(5), e62413. http://doi.org/10.1371/journal.pone.0062413

Church, J. A., Gregory, J. M., White, N. J., Platten, S. M., & Mitrovica, J. X. (2011). Understanding and projecting sea level change. *Oceanography*, *24*(2), (pp. 130–143). http://doi.org/10.5670/oceanog.2011.33.COPYRIGHT

Chust, G., Albaina, A., Aranburu, A., Borja, Á., Diekmann, O. E., Estonba, A., Franco, J., Garmendia, J. M., Iriondo, M., Muxika, I., Rendo, F., Rodríguez, J. G., Ruiz-Larrañaga, O., Serrão, E. A., & Valle, M. (2013). Connectivity, neutral theories and the assessment of species vulnerability to global change in temperate estuaries. *Estuarine, Coastal and Shelf Science*, *131*, 52–63. https://doi.org/10.1016/j.ecss.2013.08.005

Cimprich, B., & Ronis, D. L. (2003). An environmental intervention to restore attention in women with newly diagnosed breast cancer. *Cancer Nursing*, *26*(4), 284-92–4.

Cleveland, C. J., Betke, M., Federico, P., Frank, J. D., Hallam, T. G., Horn, J., Jr, J. D. L., Mccracken, G. F., Medellín, R. A., Moreno-valdez, A., Sansone, C. G., Westbrook, J. K., & Kunz, T. H. (2006). Economic value of the pest control service provided by Brazilian free-tailed bats in south-central Texas. *Frontiers in Ecology and Environment*, *4*(5), 238–243.

Cleveland, C. J., & O'Connor, P. A. (2011). Energy Return on Investment (EROI) of Oil Shale. *Sustainability*, *3*(12), (pp. 2307–2322). http://doi.org/10.3390/su3112307

Cohen, J., Slaughter, B., Kost, M., & Albert, D. (2015). *A field guide to the natural communities of Michigan.* East Lansing, Michigan USA: Michigan State Unversity Press.

Collins, M., Chandler, R. E., Cox, P. M., Huthnance, J. M., Rougier, J., & Stephenson, D. B. (2012). Quantifying future climate change. *Nature Climate Change*, *2*(6), (pp. 403–409). http://doi.org/10.1038/nclimate1414

Cook, B. I., & Seager, R. (2013). The response of the North American Monsoon to increased greenhouse gas forcing. *Journal of Geophysical Research Atmospheres*, *118*(4), (pp. 1690–1699). http://doi.org/10.1002/jgrd.50111

Cooke, S. J., Rice, J. C., Prior, K. A., Bloom, R., Jensen, O., Browne, D. R., Donaldson, L. A., Bennett, J. R., Vermaire, J. C., & Auld, G. (2016). The Canadian context for evidence-based conservation and environmental management. *Environmental Evidence*, *5*(1), 14. https://doi.org/10.1186/s13750-016-0065-8

Costanza, R., D'Arge, R., de Groot, R., Farber, S., Grasso, M., Hannon, B., Limburg, K., Naeem, S., O'Neill, R. V., Paruelo, J., Raskin, R. G., Sutton, P., & van den Belt, M. (1997). The value of the world's ecosystem services and natural capital. *Nature*, *387*(6630), 253–260. https://doi.org/10.1038/387253a0

Costanza, R., de Groot, R., Sutton, P., van der Ploeg, S., Anderson, S. J., Kubiszewski, I., Farber, S., & Turner, R. K. (2014). Changes in the global value of ecosystem services. *Global Environmental Change*, *26*, 152–158. https://doi.org/10.1016/j.gloenvcha.2014.04.002

Cox, P. M., Betts, R. A., Collins, M., Harris, P. P., Huntingford, C., & Jones, C. D. (2004). Amazonian forest dieback under climate-carbon cycle projections for the 21st century. *Theoretical and Applied Climatology*, *78*(1–3), (pp. 137–156). http://doi.org/10.1007/s00704-004-0049-4

Cox, P. M., Pearson, D., Booth, B. B., Friedlingstein, P., Huntingford, C., Jones, C. D., & Luke, C. M. (2013). Sensitivity of tropical carbon to climate change constrained by carbon dioxide variability. *Nature*, *494*(7437), (pp. 341–344). http://doi.org/10.1038/nature11882

Cullen-Unsworth, L. C., & Unsworth, R. (2013). Seagrass Meadows, Ecosystem Services, and Sustainability. *Environment: Science and Policy for Sustainable Development*, *55*(3), (pp. 14–28).

Dahl, T. E. (2011). *Status and Trends of Wetlands in the Conterminous United States 2004 to 2009. Habitat*, (November), (pp. 3–106).

Dalla-Nora, E. L., Dutra de Aguiar, A. P., Montenegro Lapola, D., & Woltjer, G. (2014). Why have land use change models for the Amazon failed to capture the amount of deforestation over the last decade? *Land Use Policy*, *39*(January 2016), (pp. 403–411). http://doi.org/10.1016/j.landusepol.2014.02.004

Danielsen, F. (2005). *The Asian Tsunami: A Protective Role for Coastal Vegetation. Science*, *310*(5748), (pp. 643–643). http://doi.org/10.1126/science.1118387

Datta, D., Chattopadhyay, R. N., & Guha, P. (2012). Community based mangrove management: A review on status and sustainability. *Journal of Environmental Management*, *107*, (pp. 84–95). http://doi.org/10.1016/j.jenvman.2012.04.013

Davidson, E. A., de Araújo, A. C., Artaxo, P., Balch, J. K., Brown, I. F., C. Bustamante, M. M., Coe, M. T., DeFries, R. S., Keller, M., Longo, M., Munger, J. W., Schroeder, W., Soares-Filho, B. S., Souza, C. M., & Wofsy, S. C. (2012). The Amazon basin in transition. *Nature*, *481*(7381), 321–328. https://doi.org/10.1038/nature10717

Davidson, N. C. (2014). *How much wetland has the world lost? Long-term and recent trends in global wetland area. Marine and Freshwater Research, 65*(10), (pp. 934–941).

De Araújo, F. M., Ferreira, L. G., & Arantes, A. E. (2012). Distribution Patterns of Burned Areas in the Brazilian Biomes: An Analysis Based on Satellite Data for the 2002-2010 Period. *Remote Sensing*. http://doi.org/10.3390/rs4071929

De Costa, W. A. J. M. (2011). *A review of the possible impacts of climate change on forests in the humid tropics. Journal of the National Science Foundation of Sri Lanka*, *39*(4), (pp. 281). http://doi.org/10.4038/jnsfsr.v39i4.3879

Defries, R. S., Rudel, T., Uriarte, M., & Hansen, M. (2010). Deforestation driven by urban population growth and agricultural trade in the twenty-first century. *Nature Geoscience*. http://doi.org/10.1038/ngeo756

Del Toro, I., Silva, R. R., & Ellison, A. M. (2015). Predicted impacts of climatic change on ant functional diversity and distributions in eastern North American forests. *Diversity and Distributions*, *21*(7), (pp. 781–791). http://doi.org/10.1111/ddi.12331

Delworth, T. L., Zeng, F., Vecchi, G. A., Yang, X., Zhang, L., & Zhang, R. (2016). The North Atlantic Oscillation as a driving force for observed rapid Arctic sea ice change , hemispheric warming , and Atlantic tropical cyclone variability. *Nature Geoscience*, *9*, (pp. 509–512). http://doi.org/10.1038/ngeo2738

Dennison, W. C., Orth, R. J., Moore, K. A., Stevenson, J. C., Carter, V., Kollar, S., Bergstrom, P. W., & Batiuk, R. A. (1993). Assessing Water Quality with Submersed Aquatic Vegetation. *BioScience*, *43*(2), 86–94.

Dentener, F., Drevet, J., Lamarque, J. F., Bey, I., Eickhout, B., Fiore, A. M., Hauglustaine, D., Horowitz, L. W., Krol, M., Kulshrestha, U. C., Lawrence, M., Galy-Lacaux, C., Rast, S., Shindell, D., Stevenson, D., Van Noije, T., Atherton, C., Bell, N., Bergman, D., Butler, T., Cofala, J., Collins, B., Doherty, R., Ellingsen, K., Galloway, J., Gauss, M., Montanaro, V., Müller, J. F., Pitari, G., Rodriguez, J., Sanderson, M., Solmon, F., Strahan, S., Schultz, M., Sudo, K., Szopa, S., & Wild, O. (2006). Nitrogen and sulfur deposition on regional and global scales: A multimodel evaluation. *Global Biogeochemical Cycles*, *20*(4), n/a-n/a. https://doi.org/10.1029/2005GB002672

Diekmann, O. E., Coyer, J. A., Ferreira, J., Olsen, J. L., Stam, W. T., Pearson, G. A., & Serrão, E. A. (2005). Population genetics of Zostera noltii along the west Iberian coast : consequences of small population size, habitat discontinuity and near-shore currents. *Marine Ecology Progress Series*, *290*, (pp. 89–96).

Dirzo, R., & Raven, P. H. (2003). Global State of Biodiversity and Loss. *Annual Review of Environment and Resources*, *28*(1), (pp. 137–167). http://doi.org/10.1146/annurev.energy.28.050302.105532

Dixon, M. J. R., Loh, J., Davidson, N. C., Beltrame, C., Freeman, R., & Walpole, M. (2016). Tracking global change in ecosystem area: The Wetland Extent Trends index. *Biological Conservation*, *193*, (pp. 27–35). http://doi.org/10.1016/j.biocon.2015.10.023

Dolch, T., Buschbaum, C., & Reise, K. (2013). Persisting intertidal seagrass beds in the northern Wadden Sea since the 1930s. *Journal of Sea Research*, *82*, (pp. 134–141).

Dove, A. (2009). *Long-term trends in major ions and nutrients in Lake Ontario. Aquatic Ecosystem Health & Management*, *12*(3), (pp. 281–295). http://doi.org/10.1080/14634980903136388

Dove, A., & Chapra, S. C. (2015). Long-term trends of nutrients and trophic response variables for the Great Lakes. *Limnology and Oceanography*, *60*(2), (pp. 696–721). http://doi.org/10.1002/lno.10055

Duarte, C. (1995). *Submerged aquatic vegetation in relation to different nutrient regimes. Ophelia*, *41*, (pp. 87–112).

Duarte, C. M. (2002). *The future of seagrass meadows*. *Environmental Conservation*, *29*(2), (pp. 192–206). http://doi.org/10.1017/S0376892902000127

Duarte, C. M., Borum, J., Short, F. T., & Walker, D. I. (2008). Seagrass ecosystem: their global status and prospects. In N. V. C. Polunin (Ed.), *Aquatic Ecosystems: Trend and Global Prospects* (pp. 281–294). Cambridge: Cambridge University Press.

Duarte, C. M., Losada, I. J., Hendriks, I. E., Mazarrasa, I., & Marbà, N. (2013). The role of coastal plant communities for climate change mitigation and adaptation. *Nature Climate Change*, *3*(11), (pp. 961–968). http://doi.org/10.1038/nclimate1970

Duggan, J. M., Eichelberger, B. A., Ma, S., Lawler, J. J., & Ziv, G. (2015). Informing management of rare species with an approach combining scenario modeling and spatially explicit risk assessment. *Ecosystem Health and Sustainability*, *1*(6), (pp. 22). http://doi.org/10.1890/EHS14-0009.1

Ellison, A. M., & Farnsworth, E. J. (1996). Anthropogenic Disturbance of Caribbean Mangrove Ecosystems: Past Impacts, Present Trends, and Future Predictions. *Biotropica*, *28*(4), (pp. 549–565). http://doi.org/10.2307/2389096

Elmqvist, T., Fragkias, M., Goodness, J., Güneralp, B., Marcotullio, P. J., McDonald, R. I., Parnell, S., Schewenius, M., Sendstad, M., Seto, K. C., Wilkinson, C., Alberti, M., Folke, C., Frantzeskaki, N., Haase, D., Katti, M., Nagendra, H., Niemelä, J., Pickett, S. T. A., Redman, C. L., & Tidball, K. (2013). Stewardship of the biosphere in the urban era. In *Urbanization, Biodiversity and Ecosystem Services: Challenges and Opportunities* (pp. 719–746). Springer Netherlands. https://doi.org/10.1007/978-94-007-7088-1_33

Engle, V. D. (2011). Estimating the Provision of Ecosystem Services by Gulf of Mexico Coastal Wetlands. *Wetlands*, *31*(1), 179–193. https://doi.org/10.1007/s13157-010-0132-9

Wetlands. Wetlands, *31*, (pp. 179–193). http://doi.org/10.1007/s13157-010-0132-9

Environment Canada. (2016). *2020 Biodiversity Goals and Targets for Canada*.

Escobar, A. (2015). *Degrowth, postdevelopment, and transitions: a preliminary conversation. Sustainability Science*, *10*(3), (pp. 451–462). http://doi.org/10.1007/s11625-015-0297-5

Evans, T., Manire, A., de Castro, F., Brondizio, E., & McCracken, S. (2001). A dynamic model of household decision-making and parcel level landcover change in the eastern Amazon. *Ecological Modelling*, *143*, (pp. 95–113).

Food and Agriculture Organisation (FAO). (2009). Food and Agriculture Organization Statistics. Fisheries and aquaculture section, FAOSTAT, United Nations. New York

Food and Agriculture Organisation (FAO). (2015). *Global Forest Resources Assessment 2015*. Rome.

Food and Agriculture Organisation (FAO). (2017a). *The Future of food and agriculture*.

Food and Agriculture Organisation (FAO). (2017b). *The future of food and agriculture - trends and challenges*. Rome.

Fearnside, P. M. (1997). *Environmental services as a strategy for sustainable development in rural Amazonia. Ecological Economics*, *20*(1), (pp. 53–70). http://doi.org/10.1016/S0921-8009(96)00066-3

Fearnside, P. M. (2008). *Amazon forest maintenance as a source of environmental services. Anais Da Academia Brasileira de Ciencias*, *80*(1), 101–114. http://doi.org/10.1590/S0001-37652008000100006

Federal, Provincial and Territorial Governments of Canada. (2010). *Canadian Biodiversity: Ecosystem Status and Trends 2010*. Canadian Councils of Resource Ministers. Ottawa, ON. 146 p. http://www.pcap-sk.org/rsu_docs/documents/Canadian_Biodiversity_Ecosystem_Status_Trends_2010.pdf

Federico, P., Hallam, T. G., McCracken, G. F., Purucker, S. T., Grant, W. E., Correa-Sandoval, A. N., Westbrook, J. K., Medellin, R. A., Cleveland, C. J., Sansone, C. G., López, J. D., Betke, M., Moreno-Valdez, A., & Kunz, T. H. (2008). Brazilian free-tailed bats as insect pest regulators in transgenic and conventional cotton crops. *Ecological Applications*, *18*(4), 826–837. Retrieved from http://www.ncbi.nlm.nih.gov/pubmed/18536245

Feldpausch, T. R., Lloyd, J., Lewis, S. L., Brienen, R. J. W., Gloor, M., Monteagudo Mendoza, A., Lopez-Gonzalez, G., Banin, L., Abu Salim, K., Affum-Baffoe, K., Alexiades, M., Almeida, S., Amaral, I., Andrade, A., Aragão, L. E. O. C., Araujo Murakami, A., Arets, E. J. M. M., Arroyo, L., Aymard C., G. A., Baker, T. R., Bánki, O. S., Berry, N. J., Cardozo, N., Chave, J., Comiskey, J. A., Alvarez, E., de Oliveira, A., Di Fiore, A., Djagbletey, G., Domingues, T. F., Erwin, T. L., Fearnside, P. M., França, M. B., Freitas, M. A., Higuchi, N., E. Honorio C., Iida, Y., Jiménez, E., Kassim, A. R., Killeen, T. J., Laurance, W. F., Lovett, J. C., Malhi, Y., Marimon, B. S., Marimon-Junior, B. H., Lenza, E., Marshall, A. R., Mendoza, C., Metcalfe, D. J., Mitchard, E. T. A., Neill, D. A., Nelson, B. W., Nilus, R., Nogueira, E. M., Parada, A., Peh, K. S.-H., Pena Cruz, A., Peñuela, M. C., Pitman, N. C. A., Prieto, A., Quesada, C. A., Ramírez, F., Ramírez-Angulo, H., Reitsma, J. M., Rudas, A., Saiz, G., Salomão, R. P., Schwarz, M., Silva, N., Silva-Espejo, J. E., Silveira, M., Sonké, B., Stropp, J., Taedoumg, H. E., Tan, S., ter Steege, H., Terborgh, J., Torello-Raventos, M., van der Heijden, G. M. F., Vásquez, R., Vilanova, E., Vos, V. A., White, L., Willcock, S., Woell, H., & Phillips, O. L. (2012). Tree height integrated into pantropical forest biomass estimates. *Biogeosciences*, *9*(8), 3381–3403. https://doi.org/10.5194/bg-9-3381-2012

Feldpausch, T. R., Phillips, O. L., Brienen, R. J. W., Gloor, E., Lloyd, J., Lopez-Gonzalez, G., Monteagudo-Mendoza, A., Malhi, Y., Alarcón, A., Álvarez Dávila, E., Alvarez-Loayza, P., Andrade, A., Aragao, L. E. O. C., Arroyo, L., Aymard C., G. A., Baker, T. R., Baraloto, C., Barroso, J., Bonal, D., Castro, W., Chama, V., Chave, J., Domingues, T. F., Fauset, S., Groot, N., Honorio Coronado, E., Laurance, S., Laurance, W. F., Lewis, S. L., Licona, J. C., Marimon, B. S., Marimon-Junior, B. H., Mendoza Bautista, C., Neill, D. A., Oliveira, E. A., Oliveira dos Santos, C., Pallqui Camacho, N. C., Pardo-Molina, G., Prieto, A., Quesada, C. A., Ramírez, F., Ramírez-Angulo, H., Réjou-Méchain, M., Rudas, A., Saiz, G., Salomão, R. P., Silva-Espejo, J. E., Silveira, M., ter Steege, H., Stropp, J., Terborgh, J., Thomas-Caesar, R., van der Heijden, G. M. F., Vásquez Martinez, R., Vilanova, E., & Vos, V. A. (2016). Amazon forest response to repeated droughts. *Global Biogeochemical Cycles*, *30*(7), 964–982. https://doi.org/10.1002/2015GB005133

Ferreira, J., Aragao, L. E. O. C., Barlow, J., Barreto, P., Berenguer, E., Bustamante, M., Gardner, T. A., Lees, A. C., Lima, A., Louzada, J., Pardini, R., Parry, L., Peres, C. A., Pompeu, P. S., Tabarelli, M., & Zuanon, J. (2014). Brazil's environmental leadership at risk. *Science*, *346*(6210), 706–707. https://doi.org/10.1126/science.1260194

Field, C., Barros, V., Mach, K., & Mastrandrea, M. (2014). *Climate change 2014: impacts, adaptation, and vulnerability*.

Filoso, S., Martinelli, L. A., Howarth, R. W., Boyer, E. W., & Dentener, F. (2006). Human activities changing the nitrogen cycle in Brazil. In L. A. Martinelli & R. W. Howarth (Eds.), *Nitrogen Cycling in the*

Americas: Natural and Anthropogenic Influences and Controls (pp. 61–89). Dordrecht: Springer Netherlands. http://doi.org/10.1007/978-1-4020-5517-1_4

Fish, M. R., Lombana, A., & Drews., C. (2009). *Regional climate projections: climate change and marine turtles in the Wider Caribbean*. San José, Costa Rica.

Foley, J. A. (2005). *Global Consequences of Land Use. Science*, *309*(5734), (pp. 570–574). http://doi.org/10.1126/science.1111772

Foley, J. A., Defries, R., Asner, G. P., Barford, C., Bonan, G., Carpenter, S. R., Chapin, F. S., Coe, M. T., Daily, G. C., Gibbs, H. K., Helkowski, J. H., Holloway, T., Howard, E. A., Kucharik, C. J., Monfreda, C., Patz, J. A., Prentice, I. C., Ramankutty, N., & Snyder, P. K. (2005). Global consequences of land use. *Science (New York, N.Y.)*, *309*(5734), 570–574. https://doi.org/10.1126/science.1111772

Folke, C., Hahn, T., Olsson, P., & Norberg, J. (2005). ADAPTIVE GOVERNANCE OF SOCIAL-ECOLOGICAL SYSTEMS. *Annual Review of Environment and Resources*, *30*, (pp. 441–473).

Ford, M. A., Cahoon, D., & Lynch, J. (1999). Restoring marsh elevation in a rapidly subsiding salt marsh by thin layer deposition of dredged material. *Ecological Engineering*, *12*, (pp. 189–205).

Fourqurean, J. W., Duarte, C. M., Kennedy, H., Marbà, N., Holmer, M., Mateo, M. A., Apostolaki, E. T., Kendrick, G. A. G. a., Krause-jensen, D., McGlathery, K. J., & Serrano, O. (2012). Seagrass ecosystems as a globally significant carbon stock. *Nature Geoscience*, *5*(7), 505–509. https://doi.org/10.1038/ngeo1477

Fragkias, M., Langanke, T., Boone, C. G., Haase, D., Marcotullio, P. J., Munroe, D., Olah, B., Reenberg, A., Seto, K. C., & Simon, D. (2012). *Land teleconnections in an Urbanizing World – A workshop report. GLP Report No. 5; GLP-IPO, Copenhagen. UGEC Report No. 6; UGEC-IPO, Tempe*.

Frick, W. F., Baerwald, E. F., Pollock, J. F., Barclay, R. M. R., Szymanski, J. A., Weller, T. J., Russell, A. L., Loeb, S. C., Medellin, R. A., & McGuire, L. P. (2017). Fatalities at wind turbines may threaten population viability of a migratory bat. *Biological Conservation*, *209*(209), 172–177. https://doi.org/10.1016/j.biocon.2017.02.023

Gadda, T., & Gasparatos, A. (2009). Land use and cover change in Japan and Tokyo's appetite for meat. *Sustainability Science*, *4*(4), (pp. 165–177).

Gagnon-Legare, A., & Prestre, P. Le. (2014). Explaining Variations in the Subnational Implementation of Global Agreements : The Case of Ecuador and the Convention on Biological Diversity. *Journal of Environment and Development*, *23*(2), (pp. 220–246). http://doi.org/10.1177/1070496514525404

Galatowitsch, S., Frelich, L., & Phillips-Mao, L. (2009). Regional climate change adaptation strategies for biodiversity conservation in a midcontinental region of North America. *Biological Conservation*, *142*(10), (pp. 2012–2022). http://doi.org/10.1016/j.biocon.2009.03.030

Gallopín, G. C., & Raskin, P. D. (2003). *Global sustainability: Bending the curve.* Routledge.

Gallopin, G. C., & Rijsberman, F. (1997). Three global water scenarios. *Int. J. Water*, *1*(1), (pp. 16–40).

Garfin, G., Franco, G., Blanco, H., Comrie, A., Gonzalez, P., Piechota, T., Smyth, R., & Waskom, R. (2014). Ch. 20: Southwest. Climate Change Impacts in the United States: The Third National Climate Assessment. In and G. W. Y. J. M. Melillo, Terese (T.C.) Richmond (Ed.), *U.S. Global Change Research Program* (pp. 462–486). USGCRP.

Gatti, L. V., Gloor, M., Miller, J. B., Doughty, C. E., Malhi, Y., Domingues, L. G., Basso, L. S., Martinewski, A., Correia, C. S. C., Borges, V. F., Freitas, S., Braz, R., Anderson, L. O., Rocha, H., Grace, J., Phillips, O. L., & Lloyd, J. (2014). Drought sensitivity of Amazonian carbon balance revealed by atmospheric measurements. *Nature*, *506*(7486), 76–80. https://doi.org/10.1038/nature12957

Giannini, T. C., Tambosi, L. R., Acosta, A. L., Jaffé, R., Saraiva, A. M., Imperatriz-Fonseca, V. L., & Metzger, J. P. (2015). Safeguarding Ecosystem Services: A Methodological Framework to Buffer the Joint Effect of Habitat Configuration and Climate Change. *PloS One*, *10*(6), e0129225. http://doi.org/10.1371/journal.pone.0129225

Giovanelli, J. G. R., Haddad, C. F. B., & Alexandrino, J. (2008). Predicting the potential distribution of the alien invasive American bullfrog (Lithobates catesbeianus) in Brazil. *Biological Invasions*, *10*(5), (pp. 585–590). http://doi.org/10.1007/s10530-007-9154-5

Gloor, M., Barichivich, J., Ziv, G., Brienen, R., Schöngart, J., Peylin, P., Ladvocat Cintra, B. B., Feldpausch, T., Phillips, O., & Baker, J. (2015). Recent Amazon climate as background for possible ongoing and future changes of Amazon humid forests. *Global Biogeochemical Cycles*, *29*(9), 1384–1399.

https://doi.org/10.1002/2014GB005080
Godar, J., Gardner, T. A., Tizado, E. J., & Pacheco, P. (2015). Correction for Godar et al., Actor-specific contributions to the deforestation slowdown in the Brazilian Amazon. *Proceedings of the National Academy of Sciences*, *112*(23), E3089–E3089. http://doi.org/10.1073/pnas.1508418112
Gonzalez-Bernat, M. J., & Clifton, J. (2017). "Living with our backs to the sea": A critical analysis of marine and coastal governance in Guatemala. *Marine Policy*, *81*(March), (pp. 9–20). http://doi.org/10.1016/j.marpol.2017.03.003
Gorell, J. M., Peterson, E. L., Rybicki, B. A., & Johnson, C. C. (2004). Multiple risk factors for Parkinson's disease. *Journal of the Neurological Sciences*, *217*(2), (pp. 169–174). http://doi.org/10.1016/j.jns.2003.09.014
Graesser, J., Aide, T. M., Grau, H. R., & Ramankutty, N. (2015). Cropland/pastureland dynamics and the slowdown of deforestation in Latin America. *Environmental Research Letters*, *10*(3), (pp. 1748–9326). http://doi.org/10.1088/1748-9326/10/3/034017
Green, E. P., & Short, F. T. (2003). *World atlas of seagrasses. Prepared by the UNEP World Conservation Monitoring Centre. Botanica Marina* (Vol. 47). Berkeley. USA: University of California Press, 298 pp. http://doi.org/10.1515/BOT.2004.029
Greve, T. M., & Binzer, T. (2004). Which factors regulate seagrass growth and distribution? In J. Borum, C. M. Duarte, D. Krause-Jensen, & T. M. Greve (Eds.), *European seagrasses: an introduction to monitoring and management* (pp. 19–23). EU project Monitoring and Managing of European Seagrasses.
Gudynas, E. (2011). *Buen Vivir: today's tomorrow. Development*, *54*(4), (pp. 441–447).
Güneralp, B., Seto, K. C., & Ramachandran, M. (2013). Evidence of urban land teleconnections and impacts on hinterlands. *Current Opinion in Environmental Sustainability*, *5*(5), (pp. 445–451). http://doi.org/10.1016/j.cosust.2013.08.003
Gunn, A., Russell, D., & Eamer, J. (2011). *Northern caribou population trends in Canada*.
Habitat III, U. (2016). *A new urban agenda. Quito declaration on sustainable cities and human settlements for all.* Quito.
Halpern, B. S., Walbridge, S., Selkoe, K. A., Kappel, C. V., Micheli, F., D'Agrosa, C., Bruno, J. F., Casey, K. S., Ebert, C., Fox, H. E., Fujita, R., Heinemann, D., Lenihan, H. S., Madin, E. M. P., Perry, M. T., Selig, E. R., Spalding, M., Steneck, R., & Watson, R. (2008). A Global Map of Human Impact on Marine Ecosystems. *Science*, *319*(5865), 948–952. https://doi.org/10.1126/science.1149345
Harley, C. D. G., & Paine, R. T. (2009). Contingencies and compounded rare perturbations dictate sudden distributional shifts during periods of gradual climate change. *Proceedings of the National Academy of Sciences of the United States of America*, *106*, (pp. 11172–11176). http://doi.org/10.1073/pnas.0904946106
Harsch, M. A., Hulme, P. E., McGlone, M. S., & Duncan, R. P. (2009). Are treelines advancing? A global meta-analysis of treeline response to climate warming. *Ecology Letters*, *12*(10), (pp. 1040–1049).
Hartig, J. H., Zarull, M. A., Ciborowski, J. J. H., Gannon, J. E., Wilke, E., Norwood, G., & Vincent, A. N. (2009). Long-term ecosystem monitoring and assessment of the Detroit River and Western Lake Erie. *Environmental Monitoring and Assessment*, *158*(1–4), (pp. 87–104). http://doi.org/10.1007/s10661-008-0567-0
Hecht, S. B. (2014). *Forests lost and found in tropical Latin America: the woodland "green revolution." The Journal of Peasant Studies*, *41*(5), (pp. 877–909). http://doi.org/10.1080/03066150.2014.917371
Hemminga, M. A., & Duarte, C. M. (2000). *Seagrass ecology*. *Psychodynamic psychiatry* (Vol. 42). Cambridge University Press. http://doi.org/10.1521/pdps.2014.42.2.319
Hill, M., & Clérici, C. (2013). *Implementación de planes de uso y manejo responsable del suelo. III Simposio nacional de agricultura 2013*.
Hootsmans, M. J. M., Vermaat, J. E., & Van Vierssen, W. (1987). Seed-bank development, germination and early seedling survival of two seagrass species from The Netherlands: Zostera marina L. and Zostera noltii hornem. *Aquatic Botany*, *28*(3–4), (pp. 275–285). http://doi.org/10.1016/0304-3770(87)90005-2
Hope, A. G., Waltari, E., Malaney, J. L., Payer, D. C., Cook, J. A., & Talbot, S. L. (2015). Arctic biodiversity: increasing richness accompanies shrinking refugia for a cold-associated tundra fauna. *Ecosphere*,

6(9), art159. http://doi.org/10.1890/ES15-00104.1

Howes, M., Wortley, L., Potts, R., Dedekorkut-Howes, A., Serrao-Neumann, S., Davidson, J., Smith, T., & Nunn, P. (2017). Environmental Sustainability: A Case of Policy Implementation Failure? *Sustainability*, *9*(2), 165. https://doi.org/10.3390/su9020165

Hu, F. S., Higuera, P. E., Duffy, P., Chipman, M. L., Rocha, A. V, Young, A. M., Kelly, R., & Dietze, M. C. (2015). Arctic tundra fires: natural variability and responses to climate change. *Frontiers in Ecology and the Environment*, *13*(7), 369–377. https://doi.org/10.1890/150063

Hu, F. S., Higuera, P. E., Walsh, J. E., Chapman, W. L., Duffy, P. A., Brubaker, L. B., & Chipman, M. L. (2010). Tundra burning in Alaska: Linkages to climatic change and sea ice retreat. *Journal of Geophysical Research*, *115*(G4), G04002. http://doi.org/10.1029/2009JG001270

Hu, Y., Chang, X., Lin, X., Wang, Y., Wang, S., Duan, J., Zhang, Z., Yang, X., Luo, C., Xu, G., & Zhao, X. (2010). Effects of warming and grazing on N2O fluxes in an alpine meadow ecosystem on the Tibetan plateau. *Soil Biology and Biochemistry*, *42*(6), 944–952. https://doi.org/10.1016/J.SOILBIO.2010.02.011

Huang, S., Yeh, C. J., & Chang, F. L. (2010). The transition to an urbanizing world and the demand for natural resources. *Environmental Sustainability*, *2*(2), (pp. 136–143).

Hughes, R. G., Williams, S. L., Duarte, C. M., Heck, K. L., & Waycott, M. (2009). Associations of concern: declining seagrasses and threatened dependent species. *Frontiers in Ecology and the Environment*, *7*(5), (pp. 242–246). http://doi.org/10.1890/080041

Hunt, D. V. L., Lombardi, D. R., Atkinson, S., Barber, A. R. G., Barnes, M., Boyko, C. T., Brown, J., Bryson, J., Butler, D., Caputo, S., Caserio, M., Coles, R., Cooper, R. F. D., Farmani, R., Gaterell, M., Hale, J., Hales, C., Hewitt, C. N., Jankovic, L., Jefferson, I., Leach, J., MacKenzie, A. R., Memon, F. A., Sadler, J. P., Weingaertner, C., Whyatt, J. D., & Rogers, C. D. F. (2012). Scenario Archetypes: Converging Rather than Diverging Themes. *Sustainability*, *4*(12), 740–772. https://doi.org/10.3390/su4040740

Huntingford, C., Zelazowski, P., Galbraith, D., Mercado, L. M., Sitch, S., Fisher, R., Lomas, M., Walker, A. P., Jones, C. D., Booth, B. B. B., Malhi, Y., Hemming, D., Kay, G., Good, P., Lewis, S. L., Phillips, O. L., Atkin, O. K., Lloyd, J., Gloor, E., Zaragoza-Castells, J., Meir, P., Betts, R., Harris, P. P., Nobre, C., Marengo, J., & Cox, P. M. (2013). Simulated resilience of tropical rainforests to CO2-induced climate change. *Nature Geoscience*, *6*(4), 268–273. https://doi.org/10.1038/ngeo1741

Huntington, H. P., Boyle, M., Flowers, G. E., Weatherly, J. W., Hamilton, L. C., Hinzman, L., Gerlach, C., Zulueta, R., Nicolson, C., & Overpeck, J. (2007). The influence of human activity in the Arctic on climate and climate impacts. *Climatic Change*, *82*(1–2), 77–92. https://doi.org/10.1007/s10584-006-9162-y

Inniss, L., & Simcock, A. (2016). *The First Global Integrated Marine Assessment: World Ocean Assessment I*.

INPE. (2014). *Sistema DEGRAD*. São José dos Campos, SP, Brazil.

INPE. (2015). *Terra Class*. São José dos Campos, São Paulo, SP, Brazil.

INPE. (2017). *Projeto PRODES: Monitoramento da Floresta Amazônica Brasileira por Satélite*. São José dos Campos, SP, Brazil.

International Joint Commission. (2016). *Protection of the Waters of the Great Lakes: 2015 Review of the Recommendations in the February 2000 report*.

IPBES. (2016). *The methodological assessment report on scenarios and models of biodiversity and ecosystem services*. Bonn, Germany.

IPCC. (2013a). *Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovern- mental Panel on Climate Change*. [Stocker, T.F., D. Qin, G.-K. Plattner, M. Tignor, S.K. Allen, J. Boschung, A. Nauels, Y. Xia, V. Bex and P.M. Midgley (eds.)]. Cambridge, United Kingdom and New York, NY, USA: Cambridge University Press.

IPCC. (2013b). *Summary for policymakers. In: Stocker, T.F., Qin, D., Plattner, G.K., Tignor, M., Allen, S.K., Boschung, J., Nauels, A., Xia, Y., Bex, V., Midgley, P.M. (Eds.), Climate Change 2013: The Physical Science Basis. Contribution of Working Group I*.

IPCC. (2014). S*ummary for Policymakers. In C. et. al Field (Ed.), Climate Change 2014: Impacts, Adaptation, and Vulnerability. Contribution of Working Group II to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change*. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Jarnevich, C., Young, N., Sheffels, T., Carter, J., Sytsma, M., & Talbert, C. (2017). Evaluating simplistic methods to understand current distributions and forecast distribution changes under climate change scenarios: an example with coypu (Myocastor coypus). *NeoBiota*, *32*, (pp. 107–125). http://doi.org/10.3897/neobiota.32.8884

Jeffries, M. O., Richter-Menge, J. A., & Overland, J. E. (Eds. . (2013). Arctic Report Card 2013.

Johnson, W. C., Millett, B. V, Gilmanov, T., Voldseth, R. A., Guntenspergen, G. R., & Naugle, D. E. (2005). Vulnerability of Northern Prairie Wetlands to Climate Change. *Bioscience*, *55*(10), (pp. 863–872). http://doi.org/10.1641/0006-3568

Jordà, G., Marbà, N., & Duarte, C. M. (2012a). Mediterranean seagrass vulnerable to regional climate warming. *Nature Climate Change*, *2*, (pp. 821–824). http://doi.org/10.1038/NCLIMATE1533

Jordà, G., Marbà, N., & Duarte, C. M. (2012b). Mediterranean seagrass vulnerable to regional climate warming. *Nature Climate Change*, *2*(11), (pp. 821–824). http://doi.org/10.1038/nclimate1533

Joshi, D. K., Hughes, B. B., & Sisk, T. D. (2015). Improving Governance for the Post-2015 Sustainable Development Goals: Scenario Forecasting the Next 50years. *World Development*, *70*, (pp. 286–302). http://doi.org/10.1016/j.worlddev.2015.01.013

Junk, W. J. (2013). *Current state of knowledge regarding South America wetlands and their future under global climate change. Aquatic Sciences*, *75*, (pp. 113–131).

Kaplan, J. O., & New, M. (2006). Arctic climate change with a 2 ∘C global warming: Timing, climate patterns and vegetation change. *Climatic Change*, *79*(3–4), (pp. 213–241). http://doi.org/10.1007/s10584-006-9113-7

Keenan, R. J., Reams, G. A., Achard, F., de Freitas, J. V., Grainger, A., & Lindquist, E. (2015). Dynamics of global forest area: Results from the FAO Global Forest Resources Assessment 2015. *Forest Ecology and Management*, *352*, (pp. 9–20). http://doi.org/10.1016/j.foreco.2015.06.014

Kirkman, H. (1997). *Seagrasses of Australia* (Australia: State of the Environment Technical Paper Series (Estuaries and the Sea)).

Klatt, B. J., & Gehring, J. L. (2013). *Assessing Bat Community Structure in Riparian and Agricultural Habitats in a High Wind Resource Area of Southeast Michigan – A Preliminary Analysis - Report Number 2013-05*. Lansing, MI.

Knowlton, N. (2001). The future of coral reefs. *Proceedings of the National Academy of Sciences of the United States of America*, *98*(10), (pp. 5419–25). http://doi.org/10.1073/pnas.091092998

Koch, M., Bowes, G., Ross, C., & Zhang, X.-H. (2013). Climate change and ocean acidification effects on seagrasses and marine macroalgae. *Global Change Biology*, *19*(1), (pp. 103–132). http://doi.org/10.1111/j.1365-2486.2012.02791.x

Kok, M. T. J., Kok, K., Peterson, G. D., Hill, R., Agard, J., & Carpenter, S. R. (2017). Biodiversity and ecosystem services require IPBES to take novel approach to scenarios. *Sustainability Science*, *12*(1), (pp. 177–181). http://doi.org/10.1007/s11625-016-0354-8

Krause-Jensen, D., Sagert, S., Schubert, H., & Boström, C. (2008). Empirical relationships linking distribution and abundance of marine vegetation to eutrophication. *Ecological Indicators*, *8*(5), (pp. 515–529). http://doi.org/10.1016/j.ecolind.2007.06.004

Kremen, C. (2015). *Reframing the land-sparing/land-sharing debate for biodiversity conservation. Annals of the New York Academy of Sciences*, *1355*(1), (pp. 52–76). http://doi.org/10.1111/nyas.12845

Kubiszewski, I., Costanza, R., Anderson, S., & Sutton, P. (2017a). The future value of ecosystem services: Global scenarios and national implications. *Ecosystem Services*, *26*, (pp. 289–301). http://doi.org/10.1016/j.ecoser.2017.05.004

Kubiszewski, I., Costanza, R., Anderson, S., & Sutton, P. (2017b). The future value of ecosystem services: Global scenarios and national implications. *Ecosystem Services*, *26*, (pp. 289–301). http://doi.org/10.1016/j.ecoser.2017.05.004

Kunz, T. H., Arnett, E. B., Cooper, B. M., Erickson, W. P., Larkin, R. P., Mabee, T., Morrison, M. L., Strickland, M. D., & Szewczak, J. M. (2007). Assessing Impacts of Wind-Energy Development on Nocturnally Active Birds and Bats: A Guidance Document. *Journal of Wildlife Management*, *71*(8), 2449–2486. https://doi.org/10.2193/2007-270

Laboratory, E. (1987). *Corps of Engineers Wetlands Delineation Manual* (Vol. 1). Vicksburg, Mississippi USA.

Lambin, E. F., Gibbs, H. K., Ferreira, L., Grau, R., Mayaux, P., Meyfroidt, P., Morton, D. C., Rudel, T. K., Gasparri, I., & Munger, J. (2013). Estimating the world's potentially available cropland using a bottom-up approach. *Global Environmental Change*, *23*(5), 892–901. https://doi.org/10.1016/J.GLOENVCHA.2013.05.005

Lapola, D. M., Martinelli, L. A., Peres, C. A., Ometto, J. P. H. B., Ferreira, M. E., Nobre, C. A., Aguiar, A. P. D., Bustamante, M. M. C., Cardoso, M. F., Costa, M. H., Joly, C. A., Leite, C. C., Moutinho, P., Sampaio, G., Strassburg, B. B. N., & Vieira, I. C. G. (2013). Pervasive transition of the Brazilian land-use system. *Nature Climate Change*, *4*(1), 27–35. https://doi.org/10.1038/nclimate2056

Lara, M. J., Johnson, D. R., Andresen, C., Hollister, R. D., & Tweedie, C. E. (2017). Peak season carbon exchange shifts from a sink to a source following 50+ years of herbivore exclusion in an Arctic tundra ecosystem. *Journal of Ecology*, *105*(1), (pp. 122–131).

Lara, L. B. L., Artaxo, P., Martinelli, L., Victoria, R., Camargo, P., Krusche, A., Ayers, G., Ferraz, E. S., & Ballester, M. (2001). Chemical composition of rainwater and anthropogenic influences in the Piracicaba River Basin, Southeast Brazil. *Atmospheric Environment*, *35*(29), 4937–4945. https://doi.org/10.1016/S1352-2310(01)00198-4

Latawiec, A. E., Strassburg, B. B. N., Rodriguez, A. M., Matt, E., Nijbroek, R., & Silos, M. (2014). Suriname: Reconciling agricultural development and conservation of unique natural wealth. *Land Use Policy*, *38*(38), (pp. 627–636). http://doi.org/10.1016/j.landusepol.2014.01.007

Laterra, P., Orúe, M. E., & Booman, G. C. (2012). Spatial complexity and ecosystem services in rural landscapes. *Agriculture, Ecosystems & Environment*, *154*, (pp. 56–67). http://doi.org/10.1016/j.agee.2011.05.013

Laurance, W. F., Oliveira, A. A., Laurance, S. G., Condit, R., Nascimento, H. E. M., Sanchez-Thorin, A. C., Lovejoy, T. E., Andrade, A., D'Angelo, S., Ribeiro, J. E., & Dick, C. W. (2004). Pervasive alteration of tree communities in undisturbed Amazonian forests. *Nature*, *428*(6979), 171–175. https://doi.org/10.1038/nature02383

Lawler, J. J., Lewis, D. J., Nelson, E., Plantinga, A. J., Polasky, S., Withey, J. C., Helmers, D. P., Martinuzzi, S., Pennington, D., & Radeloff, V. C. (2014). Projected land-use change impacts on ecosystem services in the United States. *Proceedings of the National Academy of Sciences of the United States of America*, *111*(20), 7492–7497. https://doi.org/10.1073/pnas.1405557111

Lawrence, D., & Vandecar, K. (2014). Effects of tropical deforestation on climate and agriculture. *Nature Climate Change*, *5*(1), (pp. 27–36). http://doi.org/10.1038/nclimate2430

Leadley, P., Pereira, H. M., Alkemade, R., Fernandez-Manjarrés, J. F., Proença, V., Scharlemann, J. P. W., & Walpole, M. J. (2010). *Biodiversity Scenarios: Projections of 21st century change in biodiversity and associated ecosystem services. CBD Technical Series No. 50*. Montreal.

Lenton, T. M. (2012). *Arctic Climate Tipping Points. AMBIO*, *41*(1), (pp. 10–22). http://doi.org/10.1007/s13280-011-0221-x

Les, D. H. D., Cleland, M., & Waycott, M. (1997). Phylogenetic studies in Alismatidae, II: Evolution of Marine Angiosperms (seagrasses) and Hydrophily. *Systematic Botany*, *22*, (pp. 443–463).

Lewis, S. L., Malhi, Y., & Phillips, O. L. (2004). Fingerprinting the impacts of global change on tropical forests. *Philosophical Transactions of the Royal Society B: Biological Sciences*, *359*(1443), (pp. 437–462). http://doi.org/10.1098/rstb.2003.1432

Li, J., Luo, Y., Natali, S., Schuur, E. A. G., Xia, J., Kowalczyk, E., & Wang, Y. (2014). Modeling permafrost thaw and ecosystem carbon cycle under annual and seasonal warming at an Arctic tundra site in Alaska. *Journal of Geophysical Research: Biogeosciences*. http://doi.org/10.1002/2013JG002569

Liu, J., Hull, V., Batistella, M., DeFries, R., Dietz, T., Fu, F., Hertel, T. W., Izaurralde, R. C., Lambin, E. F., Li, S., Martinelli, L. A., McConnell, W. J., Moran, E. F., Naylor, R., Ouyang, Z., Polenske, K. R., Reenberg, A., de Miranda Rocha, G., Simmons, C. S., Verburg, P. H., Vitousek, P. M., Zhang, F., & Zhu, C. (2013). Framing Sustainability in a Telecoupled World. *Ecology and Society*, *18*(2), art26. https://doi.org/10.5751/ES-05873-180226

Liu, J., Hull, V., Luo, J., Yang, W., Liu, W., Vi??a, A., Vogt, C., Xu, Z., Yang, H., Zhang, J., An, L., Chen, X., Li, S., Ouyang, Z., Xu, W., & Zhang, H. (2015). Multiple telecouplings and their complex interrelationships. *Ecology and Society*, *20*(3), 44. https://doi.org/10.5751/ES-07868-200344

Lloyd, A. H., Yoshikawa, K., Fastie, C. L., Hinzman, L., & Fraver, M. (2003). Effects of permafrost degradation on woody vegetation at arctic treeline on the Seward Peninsula, Alaska. *Permafrost and Periglacial*

Processes, *14*(2), (pp. 93–101). http://doi.org/10.1002/ppp.446

Longo, K. M., Freitas, S. R., Andreae, M. O., Yokelson, R., & Artaxo, P. (2009). Biomass Burning in Amazonia: Emissions, Long-Range Transport of Smoke and Its Regional and Remote Impacts. In M. Keller, M. Bustamante, J. Gash, & P. Silva Dias (Eds.), *Amazonia and Global Change* (pp. 207–232). American Geophysical Union. http://doi.org/10.1029/2008GM000717

Louv, R. (2008). *Last child in the woods: saving our children from nature-deficit disorder*. Chapel Hill, North Carolina, USA: Algonquin Books of Chapel Hill.

Lubchenco, J., Navarrete, S. A., Tissot, B. N., & Castilla, J. C. (1993). Possible ecological responses to global climate change: nearshore benthis biota of northeastern Pacific coastal ecosystems. In *Earth System Responses to Global Change* (pp. 147–166). San Diego, CA: Academic Press.

Macarthur, R., & Wilson, E. O. (1967). *Theory of island biogeography*. Princeton, New Jersey USA: Princeton University Press.

Mack, M. C., Bret-Harte, M. S., Hollingsworth, T. N., Jandt, R. R., Schuur, E. A. G., Shaver, G. R., & Verbyla, D. L. (2011). Carbon loss from an unprecedented Arctic tundra wildfire. *Nature*, *475*(7357), (pp. 489–492). http://doi.org/10.1038/nature10283

Mack, M. C., Schuur, E. A., Bret-Harte, M. S., Shaver, G. R., & Chapin III, F. S. (2004). Ecosystem carbon storage in arctic tundra reduced by long-term nutrient fertilization. *Nature*. http://doi.org/10.1038/nature02887

Maeda, E. E., Kim, H., Aragão, L. E. O. C., Famiglietti, J. S., & Oki, T. (2015). Disruption of hydroecological equilibrium in southwest Amazon mediated by drought. *Geophysical Research Letters*, *42*(18), (pp. 7546–7553). http://doi.org/10.1002/2015GL065252

Magrin, G.O., J.A. Marengo, J.-P. Boulanger, M.S. Buckeridge, E. Castellanos, G. Poveda, F.R. Scarano & S. Vicuña (2014), *Central and South America*. In: *Climate Change 2014: Impacts, Adaptation, and Vulnerability*. Part B: Regional Aspects. Contribution of Working Group II to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change [Barros, V.R., C.B. Field, D.J. Dokken, M.D. Mastrandrea, K.J. Mach, T.E. Bilir, M. Chatterjee, K.L. Ebi, Y.O. Estrada, R.C. Genova, B. Girma, E.S. Kissel, A.N. Levy, S. MacCracken, P.R. Mastrandrea, and L.L. White (eds.)]. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA, pp. 1499-1566

Maier, H. R., Guillaume, J. H. A., Deldena, H. van, Riddell, G. A., Haasnoot, M., & Kwakkel, J. H. (2016). An uncertain future, deep uncertainty, scenarios, robustness and adaptation: How do they fit together? *Environmental Modelling & Software*, *81*(81), (pp. 154–164). http://doi.org/10.1016/J.ENVSOFT.2016.03.014

Maine, J. J., & Boyles, J. G. (2015). Bats initiate vital agroecological interactions in corn. *Proceedings of the National Academy of Sciences of the United States of America*, *112*(40), (pp. 12438–43). http://doi.org/10.1073/pnas.1505413112

Makropoulos, C., Memon, F. A., Shirley-Smith, C., & Butler, D. (2009). Futures: An exploration of scenarios for sustainable urban water management. *Water Policy*, *10*(10), (pp. 345–373).

Malhi, Y., Aragão, L. E. O. C., Galbraith, D., Huntingford, C., Fisher, R., Zelazowski, P., Sitch, S., McSweeney, C., & Meir, P. (2009). Exploring the likelihood and mechanism of a climate-change-induced dieback of the Amazon rainforest. *Proceedings of the National Academy of Sciences of the United States of America*, *106*(49), 20610–20615. https://doi.org/10.1073/pnas.0804619106

Malhi, Y., Aragão, L. E. O. C., Metcalfe, D. B., Paiva, R., Quesada, C. A., Almeida, S., Anderson, L., Brando, P., Chambers, J. Q., Da Costa, A. C. L., Hutyra, L. R., Oliveira, P., Patiño, S., Pyle, E. H., Robertson, A. L., & Teixeira, L. M. (2009). Comprehensive assessment of carbon productivity, allocation and storage in three Amazonian forests. *Global Change Biology*, *15*(5), 1255–1274. https://doi.org/10.1111/j.1365-2486.2008.01780.x

Malhi, Y., Roberts, J. T., Betts, R. A., Killeen, T. J., Li, W., & Nobre, C. A. (2008). Climate Change, Deforestation, and the Fate of the Amazon. *Science*, *319*(5860), (pp. 169–172). http://doi.org/10.1126/science.1146961

Manuel-Navarrete, D., Gallopín, G., Blanco, M., Díaz-Zorrita, M., Ferraro, D., Herzer, H., Laterra, P., Morello, J., Murmis, M., Pengue, W., Pineiro, M., Podestá, G., Satorre, E., Torrent, M., Torres, F., Viglizzo, E., Caputo, M., & Celis, A. (2005). *Análisis sistémico de la agriculturización en la pampa húmeda argentina y sus consecuencias en regiones extra- pampeanas: sostenibilidad, brechas de* *conocimiento e integración de políticas*. *Serie de Medio Ambiente y Desarrollo* (Vol. 118). Santiago de Chile.

Manuschevich, D., & Beier, C. M. (2016). Simulating land use changes under alternative policy scenarios for conservation of native forests in south-central Chile. *Land Use Policy*, *51*, (pp. 350–362). http://doi.org/10.1016/j.landusepol.2015.08.032

Marcotullio, P. J., Hughes, S., Sarzynski, A., Pincetl, S., Sanchez Peña, L., Romero-Lankao, P., Runfola, D., & Seto, K. C. (2014). Urbanization and the carbon cycle: Contributions from social science. *Earth's Future*, *2*(10), 496–514. https://doi.org/10.1002/2014EF000257

Marengo, J. A., Alves, L. M., & Torres, R. R. (2016). Regional climate change scenarios in the Brazilian Pantanal watershed. *Climate Research*, *68*, (pp. 201–213).

Marengo, J. A., Soares, W. R., Saulo, C., & Nicolini, M. (2004). Climatology of the Low-Level Jet East of the Andes as Derived from the NCEP–NCAR Reanalyses: Characteristics and Temporal Variability. *Journal of Climate*, *17*(12), (pp. 2261–2280). http://doi.org/10.1175/1520-0442(2004)017<2261:COTLJE>2.0.CO;2

Marois, D. E., & Mitsch, W. J. (2015). Coastal protection from tsunamis and cyclones provided by mangrove wetlands - A review. *International Journal of Biodiversity Science, Ecosystem Services and Management*, *11*(1), 71–83. http://doi.org/10.1080/21513732.2014.997292

Martinelli, L. A. (2012). *Ecosystem Services and Latin America and Ecosystem Services and Agricultural Production in Latin America and Caribbean*.

Martínez-Crego, B., Vergés, A., Alcoverro, T., & Romero, J. (2008). Selection of multiple seagrass indicators for environmental biomonitoring. *Marine Ecology Progress Series*, *361*, (pp. 93–109). http://doi.org/10.3354/meps07358

Massa, S. I., Arnaud-Haond, S., Pearson, G. A., & Serrão, E. A. (2009). Temperature tolerance and survival of intertidal populations of the seagrass Zostera noltii (Hornemann) in Southern Europe (Ria Formosa, Portugal). *Hydrobiologia*, *619*(1), (pp. 195–201). http://doi.org/10.1007/s10750-008-9609-4

Mastrangelo, M. E., & Laterra, P. (2015). From biophysical to social-ecological trade-offs: integrating biodiversity conservation and agricultural production in the Argentine Dry Chaco. *Ecology and Society*, *20*(1), art20. http://doi.org/10.5751/ES-07186-200120

Mayaux, P., Holmgren, P., Achard, F., Eva, H., Stibig, H.-J., & Branthomme, A. (2005). Tropical forest cover change in the 1990s and options for future monitoring. *Philosophical Transactions of the Royal Society B: Biological Sciences*, *360*(1454), (pp. 373–384). http://doi.org/10.1098/rstb.2004.1590

McKee, L. K., D., C. R., & Feller I. C. (2007). Caribbean mangroves adjust to rising sea level through biotic controls on change in soil elevation. *Global Ecol. Biogeogr.*, *16*, (pp. 545–556).

McKenney, D. W., Pedlar, J. H., Lawrence, K., Campbell, K., & Hutchinson, M. F. (2007). Potential Impacts of Climate Change on the Distribution of North American Trees. *BioScience*, *57*(11), (pp. 939). http://doi.org/10.1641/B571106

MEA (Millenium Ecosystem Assessment). (2005). *Ecosystems and Human Well-Being: Wetlands and Water*. Washington DC: World Resources Institute (WRI).

Meltofte, H. (ed. . (2013). *Arctic Biodiversity Assessment - Status and Trends in Arctic Biodiversity*. *Arctic Biodiversity Assessment*.

Meyfroidt, P., Lambin, E. F., Erb, K. H., & Hertel, T. W. (2013). Globalization of land use: Distant drivers of land change and geographic displacement of land use. *Current Opinion in Environmental Sustainability*, *5*(5), (pp. 438–444). http://doi.org/10.1016/j.cosust.2013.04.003

Michelson, A. (2008). *Temperate Grasslands of South America. Prepared for the World Temperate Grasslands Conservation Initiative Workshop*. Hohhot, China.

Mistry, J., Tschirhart, C., Verwer, C., Glastra, R., Davis, O., Jafferally, D., Haynes, L., Benjamin, R., Albert, G., Xavier, R., Bovolo, I., & Berardi, A. (2014). Our common future? Cross-scalar scenario analysis for social–ecological sustainability of the Guiana Shield, South America. *Environmental Science & Policy*, *44*, 126–148. https://doi.org/http://dx.doi.org/10.1016/j.envsci.2014.05.007

Mitsch, W. J., & Gosselink, J. G. (2007). *Wetlands* (5th ed.). New York: John Wiley and Sons, Inc.

Mittermeier, C., Lamoreux, J., & Fonseca, G. A. . (2005). *Hotspots Revisited: Earth's Biologically Richest and Most Endangered Terrestrial Ecoregions*. Washington D.C.

Moore, K. A., & Short, F. T. (2006). Zostera: biology, ecology and management. In T. Larkum, R. Orth, & C. Duarte (Eds.), *Seagrasses: Biology, Ecology and Conservation* (pp. 361–386). The Netherlands: Springer.

Moran, D., & Kanemoto, K. (2017). Identifying species threat hotspots from global supply chains. *Nature Ecology & Evolution*. http://doi.org/10.1038/s41559-016-0023

Morello, J., Matteucci, S. D., Rodríguez, A. F., & Silva, M. E. (2012). *Ecorregiones y Complejos Ecosistémicos Argentinos*. Buenos Aires, Argentina: Editorial Orientación Gráfica.

Muehlstein, L. K., Porter, D., & Short, F. T. (1991). Labyrinthula zosterae sp. nov., the causative agent of wasting disease of eelgrass, Zostera marina. *Mycologia*, *83*, (pp. 180–191).

Mulhouse, J., Hallett, M., & Collins, S. (2017). The influence of seasonal precipitation and grass competition on 20 years of forb dynamics in northern Chihuahuan Desert grassland. *Journal of Vegetation Science*, *28*, (pp. 250–259).

Müller, R., Larrea-Alcázar, D. M., Cuéllar, S., & Espinoza, S. (2014). Causas directas de la deforestación reciente (2000-2010) y modelado de dos escenarios futuros en las tierras bajas de Bolivia lowlands and modeling of future scenarios. *Ecología En Bolivia*, *49*(1), (pp. 20–34).

Myers, P., Lundrigan, B. L., Hoffman, S. M. G., Haraminac, A. P., & Seto, S. H. (2009). Climate-induced changes in the small mammal communities of the Northern Great Lakes Region. *Global Change Biology*, *15*(6), (pp. 1434–1454). http://doi.org/10.1111/j.1365-2486.2009.01846.x

NAS (National Academy of Sciences)Agencies, F. (2007). *Environmental Impacts of Wind-Energy Projects*. http://doi.org/10.17226/11935

Natali, S. M., Schuur, E. A. G., Mauritz, M., Schade, J. D., Celis, G., Crummer, K. G., Johnston, C., Krapek, J., Pegoraro, E., Salmon, V. G., & Webb, E. E. (2015). Permafrost thaw and soil moisture driving CO2 and CH4 release from upland tundra. *Journal of Geophysical Research G: Biogeosciences*. https://doi.org/10.1002/2014JG002872

http://doi.org/10.1002/2014JG002872

National Academy of Science. (2007). *Environmental Impacts of Wind-Energy Projects*.

National Research Council. (2014). Progress Toward Restoring the Everglades: The Fifth Biennial Review: 2014. Washington, DC: The National Academies Press. http://doi.org/10.17226/18809

Nelson, E., Mendoza, G., Regetz, J., Polasky, S., Tallis, H., Cameron, D. R., Chan, K. M. a, Daily, G. C., Goldstein, J., Kareiva, P. M., Lonsdorf, E., Naidoo, R., Ricketts, T. H., & Shaw, M. R. (2009). Modeling multiple ecosystem services, biodiversity conservation, commodity production, and tradeoffs at landscape scales. *Frontiers in Ecology and the Environment*, *7*(1), 4–11. https://doi.org/10.1890/080023

Nicholls, R. J., Wong, P. P., Burkett, V. R., Codignotto, J. O., Hay, J. E., McLean, R. F., Ragoonaden, S., & Woodroffe, C. D. (2007). Coastal systems and low-lying areas. In M. L. Parry, O. F. Canziani, J. P. Palutikof, P. J. van der Linden, & C. E. Hanson (Eds.), *Climate Change 2007: Impacts, Adaptation and Vulnerability.Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change* (pp. 315–356). Cambridge, UK: Cambridge University Press.

Nogueira, E. M., Yanai, A. M., Fonseca, F. O. R., & Fearnside, P. M. (2015). Carbon stock loss from deforestation through 2013 in Brazilian Amazonia. *Global Change Biology*, *21*(3), (pp. 1271–1292). http://doi.org/10.1111/gcb.12798

O'Neill, B. C., Kriegler, E., Riahi, K., Ebi, K. L., Hallegatte, S., Carter, T. R., Mathur, R., & van Vuuren, D. P. (2014). A new scenario framework for climate change research: the concept of shared socioeconomic pathways. *Climatic Change*, *122*(3), 387–400. https://doi.org/10.1007/s10584-013-0905-2

ODEPA. (2013). *Uva de mesa: se ratifica liderazgo exportador mundial de Chile.*

OECD. (2012). *OECD Environmental Outlook to 2050: The Consequences of Inaction*. *Outlook*. http://doi.org/10.1787/9789264122246-en

Oerke, E. C. (2006). C*rop losses to pests. The Journal of Agricultural Science*, *144*(1), 31. http://doi.org/10.1017/S0021859605005708

Oliveira, C. M., Auad, A. M., Mendes, S. M., & Frizzas, M. R. (2014). Crop losses and the economic impact of insect pests on Brazilian agriculture. *Crop Protection*, *56*(56), (pp. 50–54).

http://doi.org/10.1016/J.CROPRO.2013.10.022

Oliveira, H. R., & Cassemiro, F. A. S. (2013). Potential effects of climate change on the distribution of a Caatinga's frog Rhinella granulosa (Anura, Bufonidae) . *Iheringia - Serie Zoologia*. http://doi.org/10.1590/S0073-47212013000300010

Olsen, Y. S., Sánchez-Camacho, M., Marbà, N., & Duarte, C. M. (2012). Mediterranean Seagrass Growth and Demography Responses to Experimental Warming. *Estuaries and Coasts*, *35*(5), (pp. 1205–1213). http://doi.org/10.1007/s12237-012-9521-z

Olson, D. M., Dinerstein, E., Wikramanayake, E. D., Burgess, N. D., Powell, G. V. N., Underwood, E. C., D'Amico, J. A., Itoua, I., Strand, H. E., Morrison, J. C., Loucks, C. J., Allnutt, T. F., Ricketts, T. H., Kura, Y., Lamoreux, J. F., Wettengel, W. W., Hedao, P., & Kassem, K. R. (2001). Terrestrial Ecoregions of the World: A New Map of Life on Earth. *BioScience*, *51*, 933–938.

Ometto, J. P., Aguiar, A. P., Assis, T., Soler, L., Valle, P., Tejada, G., Lapola, D. M., & Meir, P. (2014). Amazon forest biomass density maps: tackling the uncertainty in carbon emission estimates. *Climatic Change*, *124*(3), 545–560. https://doi.org/10.1007/s10584-014-1058-7

Ometto, J. P., Aguiar, A. P. D., & Martinelli, L. A. (2011). Amazon deforestation in Brazil: effects, drivers and challenges. *Carbon Management*, *2*(5), (pp. 575–585). http://doi.org/10.4155/cmt.11.48

Ondiviela, B., Losada, I. J., Lara, J. L., Maza, M., Galván, C., Bouma, T. J., & van Belzen, J. (2014). The role of seagrasses in coastal protection in a changing climate. *Coastal Engineering*, *87*, (pp. 158–168). http://doi.org/10.1016/j.coastaleng.2013.11.005

Orth, R. J., Carruthers, T. J. B., Dennison, W. C., Duarte, C. M., Fourqurean, J. W., Heck, K. L., Hughes, A. R., Kendrick, G. A., Kenworthy, W. J., Olyarnik, S., Short, F. T., Waycott, M., & Williams, S. L. (2006). A Global Crisis for Seagrass Ecosystems. *BioScience*, *56*(12), 987–996.

Osland, M. J., Gonzalez, E., & C. J. Richardson. (2011). Coastal Freshwater Wetland Plant Community Response to Seasonal Drought and Flooding in Northwestern Costa Rica. *Wetlands*, *31*, (pp. 641–652).

Oswalt, S. N., & Smith, B. W. (2014). US forest resource facts and historical trends. Forest Service FS-1035 August 2014.

Oteros-Rozas, E., Martín-López, B., Daw, T. M., Bohensky, E. L., Butler, J. R. A., Hill, R., Martin-Ortega, J., Quinlan, A., Ravera, F., Ruiz-Mallén, I., Thyresson, M., Mistry, J., Palomo, I., Peterson, G. D., Plieninger, T., Waylen, K. A., Beach, D. M., Bohnet, I. C., Hamann, M., Hanspach, J., Hubacek, K., Lavorel, S., & Vilardy, S. P. (2015). Participatory scenario planning in place-based social-ecological research: insights and experiences from 23 case studies. *Ecology and Society*, *20*(4), art32. https://doi.org/10.5751/ES-07985-200432

Outeiro, L., Häussermann, V., Viddi, F., Hucke-Gaete, R., Försterra, G., Oyarzo, H., Kosiel, K., & Villasante, S. (2014). Using ecosystem services mapping for marine spatial planning in southern Chile under scenario assessment. *Ecosystem Services*, *16*(16), 341–353. https://doi.org/10.1016/j.ecoser.2015.03.004

Pan, Y., Birdsey, R. A., Fang, J., Houghton, R., Kauppi, P. E., Kurz, W. A., Phillips, O. L., Shvidenko, A., Lewis, S. L., Canadell, J. G., Ciais, P., Jackson, R. B., Pacala, S. W., McGuire, A. D., Piao, S., Rautiainen, A., Sitch, S., & Hayes, D. (2011). A Large and Persistent Carbon Sink in the World's Forests. *Science*, *333*(6045), 988–993. https://doi.org/10.1126/science.1201609

Parlee, B., Thorpe, N., & McNabb, T. (2013). *Traditional Knowledge of Barren Ground Caribou*. Edmonton.

Parmentier, F.-J. W., Christensen, T. R., Sørensen, L. L., Rysgaard, S., Mcguire, A. D., Miller, P. A., & Walker, D. A. (2013). The impact of lower sea-ice extent on Arctic greenhouse-gas exchange. *Nature Climate Change*, *3*(3), (pp. 195–202). http://doi.org/10.1038/nclimate1784

Pascual, A., & Rodriguez-Lazaro, J. (2006). Marsh development and sea level changes in the Gernika Estuary (southern Bay of Biscay): foraminifers as tidal indicators. *Scientia Marina*, *70S1*(70S1), (pp. 101–117).

Pascual, U., Balvanera, P., Díaz, S., Pataki, G., Roth, E., Stenseke, M., Watson, R. T., Başak Dessane, E., Islar, M., Kelemen, E., Maris, V., Quaas, M., Subramanian, S. M., Wittmer, H., Adlan, A., Ahn, S. E., Al-Hafedh, Y. S., Amankwah, E., Asah, S. T., Berry, P., Bilgin, A., Breslow, S. J., Bullock, C., Cáceres, D., Daly-Hassen, H., Figueroa, E., Golden, C. D., Gómez-Baggethun, E., González-Jiménez, D., Houdet, J., Keune, H., Kumar, R., Ma, K., May, P. H., Mead, A., O'Farrell, P., Pandit, R., Pengue, W., Pichis-

Madruga, R., Popa, F., Preston, S., Pacheco-Balanza, D., Saarikoski, H., Strassburg, B. B., van den Belt, M., Verma, M., Wickson, F., & Yagi, N. (2017). Valuing nature's contributions to people: the IPBES approach. *Current Opinion in Environmental Sustainability*. https://doi.org/10.1016/j.cosust.2016.12.006

Pascual, U., Palomo, I., Adams, W. M., Chan, K. M. A., Daw, T. M., Garmendia, E., Gómez-Baggethun, E., de Groot, R. S., Mace, G. M., Martín-López, B., & Phelps, J. (2017). Off-stage ecosystem service burdens: A blind spot for global sustainability. *Environmental Research Letters*, *12*(7), 075001. https://doi.org/10.1088/1748-9326/aa7392

PBL. (2012). *Roads from Rio+20. Pathways to achieve global sustainability goals by 2050*. The Hague.

PBL. (2014). *How Sectors Can Contribute to Sustainable Use and Conservation of Biodiversity. CBD Technical Series No. 79, (79)*, (pp. 232).

PBL. (2016). *The GLOBIO model. A technical description of version 3.5.* The Hague.

Pearson, R. G., Phillips, S. J., Loranty, M. M., Beck, P. S., Damoulas, T., Knight, S. J., & Goetz, S. J. (2013). Shifts in Arctic vegetation and associated feedbacks under climate change. *Nature Climate Change*, *3*(7), (pp. 673–677).

Pengue, W. (2000). *Cultivos transgénicos, ¿Hacia dónde vamos?*

Pengue, W. A. (2005). *Transgenic Crops in Argentina: The Ecological and Social Debt. Bulletin of Science, Technology & Society, 25*(4), (pp. 314–322). http://doi.org/10.1177/0270467605277290

Pengue, W., Monterroso, I., & Binimelis, R. (2009). El caso del Sorgo de Alepo (SARG) en la agricultura argentina. *Bioinvasiones Y Bioeconomia*, *1*(1).

Pereira, H. M., Leadley, P., Proença, V., Alkemade, R., Scharlemann, J. P. W., Fernandez-Manjarrés, J. F., Araújo, M. B., Balvanera, P., Biggs, R., Cheung, W. W. L., Chini, L., Cooper, H. D., Gilman, E. L., Guénette, S., Hurtt, G. C., Huntington, H. P., Mace, G. M., Oberdorff, T., Revenga, C., Rodrigues, P., Scholes, R. J., Sumaila, U. R., & Walpole, M. (2010). Scenarios for global biodiversity in the 21st century. *Science*, *330*(6010), 1496–1501. https://doi.org/10.1126/science.1196624

Peterson, G. D., Beard, T. D. J., Beisner, B. E., Bennett, E. M., Carpenter, S. R., Cumming, G. S., Dent, C. L., & Havlicek, T. D. (2003). Assessing Future Ecosystem Services : a Case Study of the Northern Highlands Lake District , Wisconsin. *Conservation Ecology*, *7*(3), 1. https://doi.org/Artn 1

Phillips, O. L., Aragão, L. E. O. C., Lewis, S. L., Fisher, J. B., Lloyd, J., López-González, G., Malhi, Y., Monteagudo, A., Peacock, J., Quesada, C. A., van der Heijden, G., Almeida, S., Amaral, I., Arroyo, L., Aymard, G., Baker, T. R., Bánki, O., Blanc, L., Bonal, D., Brando, P., Chave, J., de Oliveira, Á. C. A., Cardozo, N. D., Czimczik, C. I., Feldpausch, T. R., Freitas, M. A., Gloor, E., Higuchi, N., Jiménez, E., Lloyd, G., Meir, P., Mendoza, C., Morel, A., Neill, D. A., Nepstad, D., Patiño, S., Peñuela, M. C., Prieto, A., Ramírez, F., Schwarz, M., Silva, J., Silveira, M., Thomas, A. S., Steege, H. ter, Stropp, J., Vásquez, R., Zelazowski, P., Dávila, E. A., Andelman, S., Andrade, A., Chao, K.-J., Erwin, T., Di Fiore, A., C., E. H., Keeling, H., Killeen, T. J., Laurance, W. F., Cruz, A. P., Pitman, N. C. A., Vargas, P. N., Ramírez-Angulo, H., Rudas, A., Salamão, R., Silva, N., Terborgh, J., & Torres-Lezama, A. (2009). Drought Sensitivity of the Amazon Rainforest. *Science*, *323*(5919), 1344 LP-1347. Retrieved from http://science.sciencemag.org/content/323/5919/1344.abstract

Phillips, O. L., Baker, T. R., Arroyo, L., Higuchi, N., Killeen, T. J., Laurance, W. F., Lewis, S. L., Lloyd, J., Malhi, Y., Monteagudo, A., Neill, D. A., Nunez Vargas, P., Silva, J. N. M., Terborgh, J., Vasquez Martinez, R., Alexiades, M., Almeida, S., Brown, S., Chave, J., Comiskey, J. A., Czimczik, C. I., Di Fiore, A., Erwin, T., Kuebler, C., Laurance, S. G., Nascimento, H. E. M., Olivier, J., Palacios, W., Patino, S., Pitman, N. C. A., Quesada, C. A., Saldias, M., Torres Lezama, A., & Vinceti, B. (2004). Pattern and process in Amazon tree turnover, 1976-2001. *Philosophical Transactions of the Royal Society B: Biological Sciences*, *359*(1443), 381–407. https://doi.org/10.1098/rstb.2003.1438

Phoenix, G. K., Hicks, W. K., Cinderby, S., Kuylenstierna, J. C. I., Stock, W. D., Dentener, F. J., Giller, K. E., Austin, A. T., Lefroy, R. D. B., Gimeno, B. S., Ashmore, M. R., & Ineson, P. (2006). Atmospheric nitrogen deposition in world biodiversity hotspots: the need for a greater global perspective in assessing N deposition impacts. *Global Change Biology*, *12*(3), 470–476. https://doi.org/10.1111/j.1365-2486.2006.01104.x

Pinho, P. F., Patenaude, G., Ometto, J. P., Meir, P., Toledo, P. M., Coelho, A., & Young, C. E. F. (2014). Ecosystem protection and poverty alleviation in the tropics: Perspective from a historical evolution of policy-making in the Brazilian Amazon. *Ecosystem Services*, *8*, (pp. 97–109). http://doi.org/10.1016/j.ecoser.2014.03.002

Piquer-Rodríguez, M., Torella, S., Gavier-Pizarro, G., Volante, J., Somma, D., Ginzburg, R., & Kuemmerle, T. (2015). Effects of past and future land conversions on forest connectivity in the Argentine Chaco. *Landscape Ecology*, *30*(5), (pp. 817–833). http://doi.org/10.1007/s10980-014-0147-3

Pithan, F., & Mauritsen, T. (2014). Arctic amplification dominated by temperature feedbacks in contemporary climate models. *Nature Geoscience*. http://doi.org/10.1038/NGEO2071

Plummer, D. A., Caya, D., Frigon, A., Côté, H., Giguère, M., Paquin, D., Biner, S., Harvey, R., & de Elia, R. (2006). Climate and Climate Change over North America as Simulated by the Canadian RCM. *Journal of Climate*, *19*(13), 3112–3132. https://doi.org/10.1175/JCLI3769.1

Potter, C., Klooster, S., & Genovese, V. (2009). Carbon emissions from deforestation in the Brazilian Amazon Region. *Biogeosciences*, *6*(11), (pp. 2369–2381). http://doi.org/10.5194/bg-6-2369-2009

Poulin, M., Pellerin, S., Cimon-Morin, J., Lavallée, S., Courchesne, G., & Tendland, Y. (2016). Inefficacy of wetland legislation for conserving Quebec wetlands as revealed by mapping of recent disturbances. *Wetlands Ecology and Management*, *24*(6), (pp. 651–665). http://doi.org/10.1007/s11273-016-9494-y

Poulter, B., Frank, D. C., Hodson, E. L., & Zimmermann, N. E. (2011). Impacts of land cover and climate data selection on understanding terrestrial carbon dynamics and the CO2 airborne fraction. *Biogeosciences*, *8*(8), 2027–2036. https://doi.org/10.5194/bg-8-2027-2011

Price, D. T., Alfaro, R. I., Brown, K. J., Flannigan, M. D., Fleming, R. A., Hogg, E. H., Girardin, M. P., Lakusta, T., Johnston, M., McKenney, D. W., Pedlar, J. H., Stratton, T., Sturrock, R. N., Thompson, I. D., Trofymow, J. A., & Venier, L. A. (2013). Anticipating the consequences of climate change for Canada's boreal forest ecosystems. *Environmental Review*, *21*, 322–365.

Quinlan, A. (2012). *Using future scenarios to explore alternate governance trajectories*. *Assessing ecosystem service governance: interactions among actors in a rural watershed in eastern Ontario*. Carleton University, Ottawa, Ontario, Canada.

Quinn, J. E., Brandle, J. R., & Johnson, R. J. (2013). A farm-scale biodiversity and ecosystem services assessment tool : the healthy farm index. *International Journal of Agricultural Sustainability*, *11*(2), (pp. 176–192).

Radeloff, V. C., Nelson, E., Plantinga, A. J., Lewis, D. J., Helmers, D., Lawler, J. J., Withey, J. C., Beaudry, F., Martinuzzi, S., Butsic, V., Lonsdorf, E., White, D., & Polasky, S. (2012). Economic-based projections of future land use in the conterminous United States under alternative policy scenarios. *Ecological Applications*, *22*(3), 1036–1049. https://doi.org/10.1890/11-0306.1

Ramirez-Villegas, J., Cuesta, F., Devenish, C., Peralvo, M., Jarvis, A., & Arnillas, C. A. (2014). Using species distributions models for designing conservation strategies of Tropical Andean biodiversity under climate change. *Journal for Nature Conservation*. http://doi.org/10.1016/j.jnc.2014.03.007

Ramsar. (2006). *Wetlands : a global disappearing act.*

Rangoonwala, A., Jones, C. E., & Ramsey, E. (III). (2016). Wetland shoreline recession in the Mississippi River Delta from petroleum oiling and cyclonic storms. *Geophysical Research Letters*, *43*(11), (pp. 652–660). http://doi.org/10.1002/2016GL070624

Raskin, P., Banuri, T., Gallopin, G., Gutman, P., Hammond, A., Kates, R., & Swart, R. (2002). *Great transition: the promise of lure of the times ahead*. Boston.

Ravera, F., Hubacek, K., Reed, M., & Tarrasón, D. (2011). Learning from Experiences in Adaptive Action Research: a Critical Comparison of two Case Studies Applying Participatory Scenario Development and Modelling Approaches. *Environmental Policy and Governance*, *21*(6), (pp. 433–453). http://doi.org/10.1002/eet.585

Ravera, F., Tarrasón, D., & Simelton, E. (2011). Envisioning adaptive strategies to change: Participatory scenarios for agropastoral semiarid systems in Nicaragua. *Ecology and Society*, *16*(1).

Raynolds, M. K., Walker, D. A., Ambrosius, K. J., Brown, J., Everett, K. R., Kanevskiy, M., Kofinas, G. P., Romanovsky, V. E., Shur, Y., & Webber, P. J. (2014). Cumulative geoecological effects of 62 years of infrastructure and climate change in ice-rich permafrost landscapes, Prudhoe Bay Oilfield, Alaska. *Global Change Biology*, *20*(4), 1211–1224. https://doi.org/10.1111/gcb.12500

Richardson, A., Brown, C. J., Brander, K. M., Bruno, J. F., Buckley, L. B., Burrows, M. T., Duarte, C. M., Halpern,

B. S., Hoegh-Guldberg, O., Holding, J., Kappel, C. V., Kiessling, W., Moore, P. J., O'Connor, M. I., Pandolfi, J. M., Parmesan, C., Schoeman, D. S., Schwing, F., Sydeman, W. J., & Poloczanska, E. S. (2012). Climate change and marine life. *Biology Letters*, *8*(6), 907–909. https://doi.org/10.1098/rsbl.2012.0530

Riensche, M., Castillo, A., Flores-Díaz, A., & Maass, M. (2015). Tourism at Costalegre, Mexico: An ecosystem services-based exploration of current challenges and alternative futures. *Futures*, *66*, (pp. 70–84). http://doi.org/10.1016/j.futures.2014.12.012

Rooth, J. E., & Stevenson, J. C. (2000). Sediment deposition patterns in Phragmites australis communities: Implications for coastal areas threatened by rising sea-level. *Wetlands Ecology and Management*. http://doi.org/10.1023/a:1008444502859

Rosa, I. M. D., Pereira, H. M., Ferrier, S., Alkemade, R., Acosta, L. A., Akcakaya, H. R., Den Belder, E., Fazel, A. M., Fujimori, S., Harfoot, M., Harhash, K. A., Harrison, P. A., Hauck, J., Hendriks, R. J. J., Hernández, G., Jetz, W., Karlsson-Vinkhuyzen, S. I., Kim, H., King, N., Kok, M. T. J., Kolomytsev, G. O., Lazarova, T., Leadley, P., Lundquist, C. J., García Márquez, J., Meyer, C., Navarro, L. M., Nesshöver, C., Ngo, H. T., Ninan, K. N., Palomo, M. G., Pereira, L. M., Peterson, G. D., Pichs, R., Popp, A., Purvis, A., Ravera, F., Rondinini, C., Sathyapalan, J., Schipper, A. M., Seppelt, R., Settele, J., Sitas, N., & Van Vuuren, D. (2017). Multiscale scenarios for nature futures. *Nature Ecology and Evolution*, *1*(10), 1416–1419. https://doi.org/10.1038/s41559-017-0273-9

Rosenthal, A., Arkema, K., Verutes, G., Bood, N., Cantor, D., Fish, M., Griffin, R., & Panuncio, M. (2013). *Identification and Valuation of Adaptation Options in Coastal-Marine Ecosystems : Test case from Placencia , Belize*.

Roura-Pascual, N., & Suarez, A. (2008). The utility of species distribution models to predict the spread of invasive ants (Hymenoptera: Formicidae) and to anticipate changes in their ranges in the face of. *Myrmecol News*, *11*(11), (pp. 67–77). http://doi.org/10.1007/s10530/007-9145-6

Rubec, C. D. A., & Hanson, A. R. (2009). Wetland mitigation and compensation : Canadian experience. *Wetlands Ecology and Management*, *17*, (pp. 3–14). http://doi.org/10.1007/s11273-008-9078-6

Rudel, T. K., Schneider, L., Uriarte, M., Turner, B. L., DeFries, R., Lawrence, D., Geoghegan, J., Hecht, S., Ickowitz, A., Lambin, E. F., Birkenholtz, T., Baptista, S., & Grau, R. (2009). Agricultural intensification and changes in cultivated areas, 1970-2005. *Proceedings of the National Academy of Sciences of the United States of America*, *106*(49), 20675–20680. https://doi.org/10.1073/pnas.0812540106

Ruiz-Mallén, I., Corbera, E., Calvo-Boyero, D., & Reyes-García, V. (2015). Participatory scenarios to explore local adaptation to global change in biosphere reserves: Experiences from Bolivia and Mexico. *Environmental Science & Policy*, *54*(54), (pp. 398–408). http://doi.org/10.1016/J.ENVSCI.2015.07.027

Saatchi, S. S., Houghton, R. A., Dos Santos Alvalá, R. C., Soares, J. V., & Yu, Y. (2007). Distribution of aboveground live biomass in the Amazon basin. *Global Change Biology*, *13*(4), (pp. 816–837). http://doi.org/10.1111/j.1365-2486.2007.01323.x

Sala, O. E. (2000). *Global Biodiversity Scenarios for the Year 2100 Science*, *287*(5459), (pp. 1770–1774). http://doi.org/10.1126/science.287.5459.1770

Sampaio, G., Nobre, C., Costa, M. H., Satyamurty, P., Soares-Filho, B. S., & Cardoso, M. (2007). Regional climate change over eastern Amazonia caused by pasture and soybean cropland expansion. *Geophysical Research Letters*, *34*(17), L17709. http://doi.org/10.1029/2007GL030612

Satorre, E. (2005). *Cambios tecnológicos en la agricultura argentina actual. Ciencia Hoy*, *15*(87), (pp. 24–31).

Satterthwaite, D., McGranahan, G., & Tacoli, C. (2010). Urbanization and its implications for food and farming. *Philosophical Transactions of the Royal Society B: Biological Sciences*, *365*(1554), (pp. 2809–2820). http://doi.org/10.1098/rstb.2010.0136

Saunders, M. I., Leon, J., Phinn, S., Callaghan, D. P., O'Brien, K. R., Roelfsema, C. M., Lovelock, C. E., Lyons, M. B., & Mumby, P. J. (2013). Coastal retreat and improved water quality mitigate losses of seagrass from sea level rise. *Global Change Biology*, *19*(8), 2569–2583. https://doi.org/10.1111/gcb.12218

Scheffer, M., Hirota, M., Holmgren, M., Van Nes, E. H., & Chapin, F. S. (2012). Thresholds for boreal biome transitions. *Proceedings of the National Academy of Sciences*, *109*(52), 21384–21389. https://doi.org/10.1073/pnas.1219844110

Schmidt-Nielsen, K. (1997). *Animal physiology: adaptation and environment*. Cambridge University Press.

Schmitt-Olabisi, L. K., Kapuscinski, A. R., Johnson, K. A., Reich, P. B., Stenquist, B., & Draeger, K. J. (2010). Using Scenario Visioning and Participatory System Dynamics Modeling to Investigate the Future: Lessons from Minnesota 2050. *Sustainability*, *2010*(2), (pp. 2686–2706). http://doi.org/10.3390/su2082686

Schneider, A., Logan, K. E., & Kucharik, C. J. (2012). Impacts of Urbanization on Ecosystem Goods and Services in the U.S. Corn Belt. *Ecosystems*, *15*(4), (pp. 519–541). http://doi.org/10.1007/s10021-012-9519-1

Schneider, F., Kallis, G., & Martinez-Alier, J. (2010). Crisis or opportunity? Economic degrowth for social equity and ecological sustainability. Introduction to this special issue. *Journal of Cleaner Production*, *18*(6), (pp. 511–518). http://doi.org/10.1016/J.JCLEPRO.2010.01.014

Schuur, E. A. G., Abbott, B. W., Bowden, W. B., Brovkin, V., Camill, P., Canadell, J. G., Chanton, J. P., Chapin, F. S., Christensen, T. R., Ciais, P., Crosby, B. T., Czimczik, C. I., Grosse, G., Harden, J., Hayes, D. J., Hugelius, G., Jastrow, J. D., Jones, J. B., Kleinen, T., Koven, C. D., Krinner, G., Kuhry, P., Lawrence, D. M., McGuire, A. D., Natali, S. M., O'Donnell, J. A., Ping, C. L., Riley, W. J., Rinke, A., Romanovsky, V. E., Sannel, A. B. K., Schädel, C., Schaefer, K., Sky, J., Subin, Z. M., Tarnocai, C., Turetsky, M. R., Waldrop, M. P., Walter Anthony, K. M., Wickland, K. P., Wilson, C. J., & Zimov, S. A. (2013). Expert assessment of vulnerability of permafrost carbon to climate change. *Climatic Change*. https://doi.org/10.1007/s10584-013-0730-7

Schuur, E. A. G., McGuire, A. D., Grosse, G., Harden, J. W., Hayes, D. J., Hugelius, G., Koven, C. D., & Kuhry, P. (2015). Climate change and the permafrost carbon feedback. *Nature*, *520*(January 2016), 171–179. https://doi.org/10.1038/nature14338

Schwartzman, S., Moutinho, P., & Hamburg, S. (2012). Policy Update: Amazon deforestation and Brazil's forest code: a crossroads for climate change. *Carbon Management*, *3*(4), (pp. 341–343). http://doi.org/10.4155/cmt.12.34

Screen, J. A., & Simmonds, I. (2010). The central role of diminishing sea ice in recent Arctic temperature amplification. *Nature*. http://doi.org/10.1038/nature09051

Seghezzo, L., Volante, J. N., Paruelo, J. M., Somma, D. J., Buliubasich, E. C., Rodríguez, H. E., Gagnon, S., & Hufty, M. (2011). Native Forests and Agriculture in Salta (Argentina): Conflicting Visions of Development. *Journal of Environment and Development*, *20*(3), 251–277. https://doi.org/10.1177/1070496511416915

Seitzinger, S. P., Svedin, U., Crumley, C. L., Steffen, W., Abdullah, S. A., Alfsen, C., Broadgate, W. J., Biermann, F., Bondre, N. R., Dearing, J. A., Deutsch, L., Dhakal, S., Elmqvist, T., Farahbakhshazad, N., Gaffney, O., Haberl, H., Lavorel, S., Mbow, C., McMichael, A. J., deMorais, J. M. F., Olsson, P., Pinho, P. F., Seto, K. C., Sinclair, P., Stafford Smith, M., & Sugar, L. (2012). Planetary Stewardship in an Urbanizing World: Beyond City Limits. *AMBIO*, *41*(8), 787–794. https://doi.org/10.1007/s13280-012-0353-7

Seto, K. C., Fragkias, M., Güneralp, B., & Reilly, M. K. (2011). A Meta-Analysis of Global Urban Land Expansion. *PLoS ONE*, *6*(8), e23777. http://doi.org/10.1371/journal.pone.0023777

Seto, K. C., Guneralp, B., & Hutyra, L. R. (2012). Global forecasts of urban expansion to 2030 and direct impacts on biodiversity and carbon pools. *Proceedings of the National Academy of Sciences*, *109*(40), (pp. 16083–16088). http://doi.org/10.1073/pnas.1211658109

Seto, K. C., & Ramankutty, N. (2016). Hidden linkages between urbanization and food systems. *Science*, *352*(6288), (pp. 943–5). http://doi.org/10.1126/science.aaf7439

Seto, K. C., Reenberg, A., Boone, C. G., Fragkias, M., Haase, D., Langanke, T., Marcotullio, P., Munroe, D. K., Olah, B., & Simon, D. (2012). Urban land teleconnections and sustainability. *Proceedings of the National Academy of Sciences*, *109*(20), 7687–7692. https://doi.org/10.1073/pnas.1117622109

Shanahan, D. F., Bush, R., Gaston, K. J., Lin, B. B., Dean, J., Barber, E., & Fuller, R. A. (2016). Health Benefits from Nature Experiences Depend on Dose. *Nature: Scientific Reports*, (June), (pp. 1–10). http://doi.org/10.1038/srep28551

Shaughnessy, F. J., Gilkerson, W., Black, J. M., Ward, D. H., & Petrie, M. (2012). Predicted eelgrass response to sea level rise and its availability to foraging Black Brant in Pacific coast estuaries. *Ecological Applications*, *22*(6), (pp. 1743–1761).

Short, F. T., & Burdick, D. M. (1996). Quantifying Eelgrass Habitat Loss in Relation to Housing Development and Nitrogen Loading in Waquoit Bay, Massachusetts. *Estuaries*, *19*(3), (pp. 730). http://doi.org/10.2307/1352532

Short, F. T., Carruthers, T. J. B., Dennison, W. C., & Waycott, M. (2007). Global seagrass distribution and diversity: A bioregional model. *Journal of Experimental Marine Biology and Ecology*, *350*(1–2), (pp. 3–20). http://doi.org/10.1016/j.jembe.2007.06.012

Short, F. T., Koch, E. W., Creed, J. C., Magalhães, K. M., Fernandez, E., & Gaeckle, J. L. (2006). SeagrassNet monitoring across the Americas: case studies of seagrass decline. *Marine Ecology*, *27*(4), (pp. 277–289). http://doi.org/10.1111/j.1439-0485.2006.00095.x

Short, F. T., & Neckles, H. A. (1999). The effects of global climate change on seagrasses. *Aquatic Botany*, *63*(3–4), (pp. 169–196). http://doi.org/10.1016/S0304-3770(98)00117-X

Short, F. T., Polidoro, B., Livingstone, S. R., Carpenter, K. E., Bandeira, S., Sidik Bujang, J., Calumpong, H. P., Carruthers, T. J. B., Coles, R. G., Dennison, W. C., Erftemeijer, P. L. A., Fortes, M. D., Freeman, A. S., Jagtap, T. G., Kamal, A. H. M., Kendrick, G. A., Kenworthy, W. J., La Nafie, Y. A., Nasution, I. M., Orth, R. J., Prathep, A., Sanciangco, J. C., van Tussenbroek, B., Vergara, S. G., Waycott, M., & Zieman, J. C. (2011). Extinction risk assessment of the world's seagrass species. *Biological Conservation*, *144*(7), 1961–1971. https://doi.org/10.1016/j.biocon.2011.04.010

Sistla, S. a, Moore, J. C., Simpson, R. T., Gough, L., Shaver, G. R., & Schimel, J. P. (2013). Long-term warming restructures Arctic tundra without changing net soil carbon storage. *Nature*. http://doi.org/10.1038/nature12129

Sitch, S., David Mcguire, A., Kimball, J., Gedney, N., Gamon, J., Engstrom, R., Wolf, A., Zhuang, Q., Clein, J., & McDonald, K. C. (2007). Assessing the carbon balance of circumpolar Arctic tundra using remote sensing and process modeling. *Ecological Applications*. https://doi.org/10.1890/1051-0761(2007)017[0213:ATCBOC]2.0.CO;2

Souza, R., Feng, X., Antonino, A., Montenegro, S., Souza, E., & Porporato, A. (2016). Vegetation response to rainfall seasonality and interannual variability in tropical dry forests. *Hydrological Processes*, (30), (pp. 3583–3595).

Sparovek, G., Barretto, A. G. de O. P., Matsumoto, M., & Berndes, G. (2015). Effects of Governance on Availability of Land for Agriculture and Conservation in Brazil. *Environmental Science & Technology*, *49*(17), (pp. 10285–10293). http://doi.org/10.1021/acs.est.5b01300

Spracklen, D. V., & Garcia-Carreras, L. (2015). The impact of Amazonian deforestation on Amazon basin rainfall. *Geophysical Research Letters*, *42*(21), (pp. 9546–9552). http://doi.org/10.1002/2015GL066063

Stehfest, E., Bouwman, L., Van Vuuren, D. P., Den Elzen, M. G. J., Eickhout, B., & Kabat, P. (2009). Climate benefits of changing diet. *Climatic Change*, *95*(1–2), (pp. 83–102). http://doi.org/10.1007/s10584-008-9534-6

Stehfest, E., van Vuuren, D., Kram, T., Bouwman, L., Alkemade, R., Bakkenes, M., Biemans, H., Bouwman, A., den Elzen, M., Janse, J., Lucas, P., van Minnen, J., Muller, M., & Prins, A. G. (2014). *Integrated Assessment of Global Environmental Change with IMAGE 3.0 - Model description and policy applications*. The Hague: PBL Netherlands Environmental Assessment Agency.

Stewart, E. J., Liggett, D., & Dawson, J. (2017). The evolution of polar tourism scholarship : research themes , networks and agendas. *Polar Geography*, *40*(1), (pp. 59–84). http://doi.org/10.1080/1088937X.2016.1274789

Strickland, M. D., Arnett, E. B., Erickson, W. P., Johnson, D. H., Johnson, G. D., M.L., M., Shaffer, J. A., & Warren-Hicks, W. (2011). *Comprehensive Guide to Studying Wind Energy/Wildlife Interactions*. Washington, D.C., USA.

Sun, J., TONG, Y., & Liu, J. (2017). Telecoupled land-use changes in distant countries. *Journal of Integrative Agriculture*, *16*(2), (pp. 368–376). http://doi.org/10.1016/S2095-3119(16)61528-9

Sweet, S. K., Griffin, K. L., Steltzer, H., Gough, L., & Boelman, N. T. (2015). Greater deciduous shrub abundance extends tundra peak season and increases modeled net CO2 uptake. *Global Change Biology*. http://doi.org/10.1111/gcb.12852

Sy, V. De, Herold, M., Achard, F., Beuchle, R., Clevers, J. G. P. W., Lindquist, E., & Verchot, L. (2015). Land use patterns and related carbon losses following deforestation in South America. *Environmental Research Letters*, *10*(12), 124004. http://doi.org/10.1088/1748-9326/10/12/124004

Tarnocai, C., Canadell, J. G., Schuur, E. A. G., Kuhry, P., Mazhitova, G., & Zimov, S. (2009). Soil organic carbon pools in the northern circumpolar permafrost region. *Global Biogeochemical Cycles*, *23*(2), n/a-n/a. http://doi.org/10.1029/2008GB003327

Teh, L. S. L., Cheung, W. W. L., & Sumaila, U. R. (2016). Scenarios for investigating the future of Canada's oceans and marine fisheries under environmental and socioeconomic change. *Regional Environmental Change*. http://doi.org/10.1007/s10113-016-1081-5

Tejada, G., Dalla-Nora, E., Cordoba, D., Lafortezza, R., Ovando, A., Assis, T., & Aguiar, A. P. (2016). Deforestation scenarios for the Bolivian lowlands. *Environmental Research*, *144*, *PartB*, (pp. 49–63). http://doi.org/10.1016/j.envres.2015.10.010

Ter Steege, H., Pitman, N., Sabatier, D., Castellanos, H., Van Der Hout, P., Daly, D. C., Silveira, M., Phillips, O., Vasquez, R., Van Andel, T., Duivenvoorden, J., De Oliveira, A. A., Ek, R., Lilwah, R., Thomas, R., Van Essen, J., Baider, C., Maas, P., Mori, S., Terborgh, J., NúÑez Vargas, P., Mogollón, H., & Morawetz, W. (2003). A spatial model of tree α-diversity and tree density for the Amazon. *Biodiversity and Conservation*, *12*(11), 2255–2277. https://doi.org/10.1023/A:1024593414624

Uden, D. R., Allen, C. R. ., Bishop, A. A., Grosse, R., Jorgensen, C. F., LaGrange, T. G., Stutheit, R. G., & Vrtiska, M. P. (2015). Predictions of future ephemeral springtime waterbird stopover habitat availability under global change. *Ecosphere*, *6*(11), Article 215. https://doi.org/10.1890/ES15-00256.1

UN. (2017). *New Urban Agenda*. Quito, Ecuador.

UNEP (2010) *Assessing the Environmental Impacts of Consumption and Production: Priority Products and Materials, A Report of the Working Group on the Environmental Impacts of Products and Materials to the International Panel for Sustainable Resource Management.* Hertwich, E., van der Voet, E., Suh, S., Tukker, A, Huijbregts M., Kazmierczyk, P., Lenzen, M., McNeely, J., Moriguchi, Y. http://www.resourcepanel.org/sites/default/files/documents/document/media/assessing_scp_summary_report_english.pdf

UNEP. (2014). *Annual Report 2014*.

UNEP. (2016a). *GEO-6 Regional Assessment for Latin America and the Caribbean*. Nairobi, Kenya: United Nations Environment Programme.

UNEP. (2016b). *GEO-6 Regional Assessment for North America*. Nairobi, Kenya.

Urbina-Cardona, J. N., & Castro, F. (2010). Distribución actual y futura de anfibios y reptiles con potencial invasor en Colombia: una aproximación usando modelos de nicho ecológico. *Diversidad Y Cambio Climático*, *1*(1), (pp. 65–72). http://doi.org/10.1016/j.jns.2003.09.014

Urbina-Cardona, J. N., & Flores-Villela, O. (2010). Ecological-niche modeling and prioritization of conservation-area networks for Mexican herpetofauna. *Conservation Biology*, *24*(4), (pp. 1031–41). http://doi.org/10.1111/j.1523-1739.2009.01432.x

USDA. (2009). *World Crop Production summary*. Washington D.C., USA: Foreign Agricultural Service, United States Department of Agriculture (USDA).

USEPA. (2011). *National Wetland Condition Assessment: 2011 Technical Report. EPA-843-R-15-006*. Washington D.C.

Valle, M., Chust, G., del Campo, A., Wisz, M. S., Olsen, S. M., Garmendia, J. M., & Borja, Á. (2014). Projecting future distribution of the seagrass Zostera noltii under global warming and sea level rise. *Biological Conservation*, *170*, (pp. 74–85). http://doi.org/10.1016/j.biocon.2013.12.017

van Soesbergen, A. J. J., & Mulligan, M. (2014). Modelling multiple threats to water security in the Peruvian Amazon using the WaterWorld policy support system. *Earth System Dynamics*, *5*(1), (pp. 55–65). http://doi.org/10.5194/esd-5-55-2014

van Vuuren, D. P., Kok, M. T. J., Girod, B., Lucas, P. L., & de Vries, B. (2012). Scenarios in Global Environmental Assessments: Key characteristics and lessons for future use. *Global Environmental Change*, *22*(4), (pp. 884–895). http://doi.org/10.1016/j.gloenvcha.2012.06.001

Velasquez-Tibata, J., P. Salaman, & Graham, C. H. (2013). Effects of climate change on species distribution, community structure, and conservation of birds in protected areas in Colombia. *Regional Environmental Change*, *13*, (pp. 235–248).

Vennum, T. (1988). *Wild rice and the Ojibway people*. (M. H. S. Press, Ed.). St. Paul, Minnesota, USA.

Verutes, G. M., Huang, C., Estrella, R. R., & Loyd, K. (2014). Exploring scenarios of light pollution from coastal development reaching sea turtle nesting beaches near Cabo Pulmo, Mexico. *Global Ecology and*

Conservation, *2*(2), (pp. 170–180). http://doi.org/10.1016/j.gecco.2014.09.001

Victor, P. A. (2012). *Growth, degrowth and climate change: A scenario analysis. Ecological Economics, 84*, (pp. 206–212). http://doi.org/10.1016/j.ecolecon.2011.04.013

Viglizzo, E., Carreno, L., Volante, J. N., & Mosciaro, M. J. (2011). Valuación de los Bienes y Servicios Ecosistémicos: Verdad objetiva o cuento de la buena pipa? In P. Laterra, E. Jobbágy, & J. Paruelo (Eds.), *Valoración de Servicios Ecosistémicos, Conceptos, herramientas y aplicaciones para el ordenamiento territorial*. Ediciones INTA.

Vilardy-Quiroga, S. P., & González Novoa, J. A. (2011). *Repensando la Ciénaga: Nuevas miradas y estrategias para la sostenibilidad en la Ciénaga Grande de Santa Marta* (Vol. 1). http://doi.org/10.1017/CBO9781107415324.004

Visconti, P., Bakkenes, M., Baisero, D., Brooks, T., Butchart, S. H. M., Joppa, L., Alkemade, R., Di Marco, M., Santini, L., Hoffmann, M., Maiorano, L., Pressey, R. L., Arponen, A., Boitani, L., Reside, A. E., van Vuuren, D. P., & Rondinini, C. (2016). Projecting Global Biodiversity Indicators under Future Development Scenarios. *Conservation Letters*, *9*(1), 5–13. https://doi.org/10.1111/conl.12159

Vuille, M. (2013). *Climate Change and Water Resources in the Tropical Andes. Inter-American Development Bank Environmental Safeguards Unit*. Technical Note No. IDB-TN-515.

Walker, D. . (1991). *The effect of sea temperature on seagrasses and algae on the Western Australian coastline. Journal of the Royal Society of Western Australia*, *74*, (pp. 71–77).

Walker, M. D., Wahren, C. H., Hollister, R. D., Henry, G. H. R., Ahlquist, L. E., Alatalo, J. M., Bret-Harte, M. S., Calef, M. P., Callaghan, T. V, Carroll, A. B., Epstein, H. E., Jónsdóttir, I. S., Klein, J. A., Magnússon, B., Molau, U., Oberbauer, S. F., Rewa, S. P., Robinson, C. H., Shaver, G. R., Suding, K. N., Thompson, C. C., Tolvanen, A., Totland, Ø., Turner, P. L., Tweedie, C. E., Webber, P. J., & Wookey, P. A. (2006). Plant community responses to experimental warming across the tundra biome. *Proceedings of the National Academy of Sciences*. https://doi.org/10.1073/pnas.0503198103

Ward, J. D., Sutton, P. C., Werner, A. D., Costanza, R., Mohr, S. H., & Simmons, C. T. (2016). Is decoupling GDP growth from environmental impact possible? *PLoS ONE*, *11*(10), (pp. 1–14). http://doi.org/10.1371/journal.pone.0164733

Waycott, M., Collier, C., Mcmahon, K., Ralph, P., Mckenzie, L., Udy, J., & Grech, A. (2007). Vulnerability of seagrasses in the Great Barrier Reef to climate change. In J. E. Johnson & P. A. Marshal (Eds.), *Climate Change and the Great Barrier Reef: A Vulnerability Assessment* (pp. 193–299). Australia: Great Barrier Reef Marine Park Authority and Australian Greenhouse Office.

Waycott, M., Duarte, C. M., Carruthers, T. J. B., Orth, R. J., Dennison, W. C., Olyarnik, S., Calladine, A., Fourqurean, J. W., Heck, K. L., Hughes, R. G., Kendrick, G. A., Kenworthy, W. J., Short, F. T., & Williams, S. L. (2009). Accelerating loss of seagrasses across the globe threatens coastal ecosystems. *Proceedings of the National Academy of Sciences of the United States of America*, *106*(30), 12377–12381. https://doi.org/10.1073/pnas.0905620106

Waylen, K. A., Martin-Ortega, J., Blackstock, K. L., Brown, I., Avendaño Uribe, B. E., Basurto Hernández, S., Bertoni, M. B., Bustos, M. L., Cruz Bayer, A. X., Escalante Semerena, R. I., Farah Quijano, M. A., Ferrelli, F., Fidalgo, G. L., Hernández López, I., Huamantinco Cisneros, M. A., London, S., Maya Vélez, D. L., Ocampo-Díaz, N., Ortiz-Guerrero, C. E., Pascale, J. C., Perillo, G. M. E., Piccolo, M. C., Pinzón Martínez, L. N., Rojas, M. L., Scordo, F., Vitale, V., & Zilio, M. I. (2015). Can scenario-planning support community-based natural resource management? Experiences from three countries in Latin America. *Ecology and Society*, *20*(4), 28. https://doi.org/10.5751/ES-07926-200428

Weatherbee, E. E., & Klatt, B. (2004). *Detroit Parks: Plant surveys and community characterizations*. Pinckney, Michigan USA.

Webb, E. E., Schuur, E. A. G., Natali, S. M., Oken, K. L., Bracho, R., Krapek, J. P., Risk, D., & Nickerson, N. R. (2016). Increased wintertime CO 2 loss as a result of sustained tundra warming. *Journal of Geophysical Research: Biogeosciences*, *121*(2), 249–265. https://doi.org/10.1002/2014JG002795

Weinzettel, J., Hertwich, E. G., Peters, G. P., Steen-Olsen, K., & Galli, A. (2013). Affluence drives the global displacement of land use. *Global Environmental Change*, *23*(2), (pp. 433–438). http://doi.org/10.1016/j.gloenvcha.2012.12.010

Wernberg, T., Russell, B. D., Thomsen, M. S., Gurgel, C. F. D., Bradshaw, C. J. a, Poloczanska, E. S., & Connell, S. D. (2011). Seaweed communities in retreat from ocean warming. *Current Biology : CB*, *21*(21), (pp. 1828–32). http://doi.org/10.1016/j.cub.2011.09.028

Werth, D. (2002). *The local and global effects of Amazon deforestation. Journal of Geophysical Research*, *107*(D20), 8087. http://doi.org/10.1029/2001JD000717

West, A. M., Kumar, S., Wakie, T., Brown, C. S., Stohlgren, T. J., Laituri, M., & Bromberg, J. (2015). Using High-Resolution Future Climate Scenarios to Forecast Bromus tectorum Invasion in Rocky Mountain National Park. *PloS One*, *10*(2), e0117893. http://doi.org/10.1371/journal.pone.0117893

Willaarts, B. A., Salmoral, G., Farinaci, J., & Sanz-Sánchez, M. J. (2014). Trends in land use and ecosystem services. In B. A. Willaarts, A. Garrido, & M. R. Llamas (Eds.), *Water for Food and Wellbeing in Latin America and the Caribbean. Social and Environmental Implications for a Globalized Economy* (pp. 55–80). Oxon and New York: Routledge.

Willett, W. (2001). *Eat, drink, and be healthy: the Harvard Medical School guide to healthy eating*. New York: Simon & Schuster.

Wong, P. P., Losada, I. J., Gattusso, J. P., Hinkel, J., Khattabi, A., McInnes, K., Saito, Y., & Sallenger, A. (2014). Coastal systems and low-lying areas. In C. et al. Field (Ed.), *Climate Change 2014: Impacts, Adaptation, and Vulnerability. Contribution of Working Group II to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change* (p. in press). Cambridge; United Kingdom and New York; NY; USA: Cambridge University Press.

WWAP (United Nations World Water Assessment Programme). (2015). *The United Nations World Water Development Report 2015: Water for a sustainable world*.

WWAP (United Nations World Water Assessment Programme). (2016). *The United Nations World Water Development Report 2016: Water and Jobs*.

WWAP (United Nations World Water Assessment Programme). (2017). *The United Nations World Water Development Report 2017*.

WWF (2016). Charity, S., Dudley, N., Oliveira, D. and S. Stolton (editors). 2016. Living Amazon Report 2016: A regional approach to conservation in the Amazon. WWF Living Amazon Initiative, Brasília and Quito.

Young, J. C., Waylen, K. A., Sarkki, S., Albon, S., Bainbridge, I., Balian, E., Davidson, J., Edwards, D., Fairley, R., Margerison, C., Mccracken, D., Owen, R., Quine, C. P., Stewart-roper, C., Thompson, D., Tinch, R., Hove, S. Van Den, & Watt, A. (2014). Improving the science-policy dialogue to meet the challenges of biodiversity conservation : having conversations rather than talking at one-another. *Biodiversity Conservation*, *23*, 387–404. https://doi.org/10.1007/s10531-013-0607-0

Yu, Y., Feng, K., & Hubacek, K. (2013). Tele-connecting local consumption to global land use. *Global Environmental Change*, *23*(5), (pp. 1178–1186). http://doi.org/10.1016/j.gloenvcha.2013.04.006

Zanella, L., Tristão Borém, R. A., Gusmão Souza, C., Ramos Alves, H. M., & Meira Borem, F. (2012). Atlantic Forest Fragmentation Analysis and Landscape Restoration Management Scenarios. *Natureza & Conservação*, *10*(1), (pp. 57–63). http://doi.org/10.4322/natcon.2012.010

Zelazowski, P., Malhi, Y., Huntingford, C., Sitch, S., & Fisher, J. B. (2011). Changes in the potential distribution of humid tropical forests on a warmer planet. *Philosophical Transactions of the Royal Society A: Mathematical, Physical and Engineering Sciences*, *369*(1934), (pp. 137–160). http://doi.org/10.1098/rsta.2010.0238

Zemp, D. C., Schleussner, C.-F., Barbosa, H. M. J., Hirota, M., Montade, V., Sampaio, G., Staal, A., Wang-Erlandsson, L., & Rammig, A. (2017). Self-amplified Amazon forest loss due to vegetation-atmosphere feedbacks. *Nature Communications*, *8*, 14681. https://doi.org/10.1038/ncomms14681

Zhang, K., Liu, H., Li, Y., Xu, H., Shen, J., Rhome, J., & Smith, T. J. (2012). The role of mangroves in attenuating storm surges. *Estuarine, Coastal and Shelf Science*, *102–103*, (pp. 11–23). http://doi.org/10.1016/j.ecss.2012.02.021

Zhao, M., & Running, S. W. (2010). Drought-Induced Reduction in Global Terrestrial Net Primary Production from 2000 Through 2009. *Science*, *329*(5994), (pp. 940–943). http://doi.org/10.1126/science.1192666

Zunckel, M., Saizar, C., & Zarauz, J. (2003). Rainwater composition in northeast Uruguay. *Atmospheric Environment*, *37*(12), (pp. 1601–1611). http://doi.org/10.1016/S1352-2310(03)00007-4

Chapter 6: Options for governance and decision-making across scales and sectors

Coordinating Lead Authors:
Fabio R. Scarano (Brazil), Keisha Garcia (Trinidad and Tobago), Antonio Diaz-de-Leon (Mexico)

Lead Authors:
Helder Lima Queiroz (Brazil), Vanesa Rodríguez Osuna (Bolivia & USA), Luciana C. Silvestri (Argentina), Cristóbal F. Díaz M. (Cuba), Octavio Pérez-Maqueo (Mexico), Marina Rosales B. (Peru), Dalia M. Salabarria F. (Cuba), Ederson A. Zanetti (Brazil)

Fellow:
Juliana S. Farinaci (Brazil)

Review Editors:
Gustavo A.B. Fonseca (Brazil/USA), Laura Nahuelhual M. (Chile)

To be cited as:
Scarano, F. R., Garcia, K., Diaz-de-Leon, A., Queiroz., H. L., Rodríguez Osuna., V., Silvestri, L. C., Díaz M., C. F., Pérez-Maqueo, O., Rosales B., M., Salabarria F., D. M., Zanetti, E. A., and Farinacci, J. S. Chapter 6: Options for governance and decision-making across scales and sectors. In IPBES (2018): The IPBES regional assessment report on biodiversity and ecosystem services for the Americas. Rice, J., Seixas, C. S., Zaccagnini, M. E., Bedoya-Gaitán, M., and Valderrama, N. (eds.). Secretariat of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services, Bonn, Germany, pp. xx-xx.

Table of contents Chapter 6

6 Executive summary

1. **For most countries of the region, environment is mostly dealt with as a separate sector in national planning, and has hitherto not been effectively mainstreamed across development sectors (*well established*).** Moreover, the development pressures outpace or outweigh the development and implementation of policies that can attend to the growing drivers affecting biodiversity and ecosystem services. This is especially true for the developing countries in the Americas region; and accounts for many of the negative trends in biodiversity and ecosystem services that are evident across the region *(well established)*. For example, in Latin America and the Caribbean, natural resource use policies often come into place only when fundamental shifts in land-use are already underway such that interventions tend to become more costly and have limited influence *(established but incomplete)*. {6.1.1, 6.2, 6.3.1, 6.3.4, 6.4}

2. **Despite reported reductions in the rate of loss in specific biomes in the Americas, the net loss that is currently evident in almost every aspect of the region's natural ecosystems is expected to continue through to 2050, driven largely by unsustainable agricultural practices and climate change (*established but incomplete*).** This will result in reductions in the adaptive capacity of the societies throughout the region, especially economically vulnerable communities in Latin America and the Caribbean (*established but incomplete*). {6.1.1, 6.4, 6.6.4}

3. **There are threats to the goal of achieving a fair balance between a healthy environment and enhanced quality of life across the region.** In addition to the speed of climate and land use change, and the persistence of poverty, the region continues to be challenged by failure to implement designed policies, lack of transparency and/or accountability of key stakeholders, failure to acknowledge indigenous and local knowledge and practices, difficulty in engaging the public or developing truly participatory mechanisms for decision-making (*established but incomplete*). {6.1.1, 6.2, 6.3, 6.4, 6.5, 6.6, 6.7}

4. **There are evidences of leakage and spillover effects in many levels and scales across the region, but they remain understudied.** Cases where environmentally damaging activities are relocated elsewhere after being stopped locally are found from protected area level to biome level (*established but incomplete*). Such issues are often unforeseen either due to lack of systemic planning or adequate mapping of potential stakeholders (*inconclusive*). {6.3.4}

5. **Ecological restoration is having positive effects at local scales, speeding up ecosystem recovery in many cases (*established but incomplete*).** However, restoration of ecosystems and species has high up-front costs and usually requires long periods of time (*well established*). Furthermore, full reversal of degradation, if possible at all, has not been demonstrated (*established but incomplete*). This indicates that countries are likely to benefit from acting quickly to invest in the conservation and sustainable use of their existing ecological infrastructure. In this context, ecosystem-based strategies incorporated into national and sub-national-level planning are a gap to be filled in the region.{6.4, 6.6.3}

6. **For most countries, global goals, targets and aspirations (Sustainable Development Goals, Aichi targets, national determined contributions) are neither aligned with nor integrated into national policies *(inconclusive)*.** As a result, the rate of achievement of global commitments vary largely between countries. For instance, among the 2020 Aichi targets of the Convention on Biological Diversity, in Canada and Latin America and the Caribbean, most progress has been reported in target 11 (protected areas). In Latin America and the Caribbean, there is also reported progress on target 17 (adoption and implementation of policy instruments), target 1 (people aware of the value of biodiversity and the steps to conserve and sustainably use it),

target 16 (Nagoya Protocol) and target 19 (improved biodiversity information sharing). In Latin America and the Caribbean, the targets most lagging behind however are target 6 (anthropogenic pressures/ direct drivers of change minimized) and target 10 (management of fish and aquatic invertebrate stocks) {6.5}.

7. **There is an overall lack of policy evaluation in the Americas, which is more pronounced in Latin America and the Caribbean than it is in North America (*established but incomplete*).** Information on policy effectiveness is often derived through case studies and anecdotal accounts {6.4.1, 6.6.1, 6.7}.

8. **Participatory deliberative processes contribute to a large class of problem-solving situations and can support successful governance (*established but incomplete*).** This is evidenced by a diversity of cases across policy areas, levels of economic development, and political cultures. However, there are reports of cases when the participatory process is flawed {6.3, 6.4.1, 6.6.4}.

9. **There is use or interest in a broad array of policy instruments by a range of actors to support biodiversity and ecosystem services management, but their implementation, even when effective locally, often do not add up to overall effectiveness at national or regional scales (*inconclusive*)**. This is evidenced by the persistent, growing intensity of most driving forces, and the negative trends apparent in biodiversity and ecosystem services across the region. Types of policy instruments found in the region include conservation incentive mechanisms (e.g. Socio-Bosque in Ecuador, Bolsa Verde in Brazil; Fonafifo in Costa Rica); protected areas (e.g. the large terrestrial cover attained in the Amazon), including marine protected areas (e.g. network governance schemes in Colombia, Ecuador, Jamaica, United States of America); natural capital accounting (e.g. North America); eco-certification (managed by governments, research institutions, non-governmental organizations, multiple stakeholders or individual companies across the region); and biodiversity offsets (mainly in North America) {6.4}.

10. **Indigenous peoples throughout the Americas have developed many different socioecological and governance systems (nationally and locally), which exist in parallel to mainstream governance (*well established*).** Although conflicts persist both in 15 countries that formally acknowledge such rights and 20 countries that do not, indigenous and local knowledge and practices are expressions of social capital that can positively influence biodiversity and ecosystem services (*established but incomplete*) {6.3.5, 6.4.1.1, 6.4.3}.

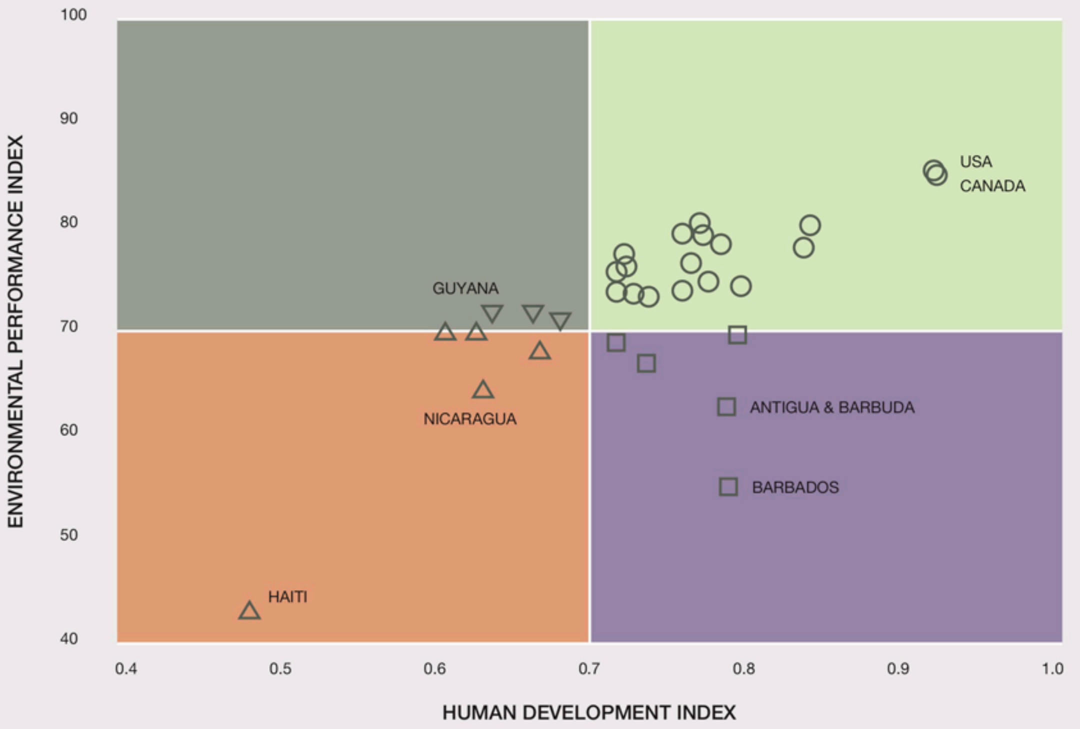
6.1 Setting the scene

6.1.1 Americas in context

In parallel to holding the largest biological diversity on the planet and several of the world's megadiverse countries, the Americas display a mosaic of socioeconomic conditions, cultures, and political systems, as well as countries that range from island-size to continental-size (Chapters 1, 2). The region's diversity is also evident in the fact that the Americas spans the full spectrum both of human development and of environmental performance - from highest to lowest (Martins et al., 2006) – as illustrated in Figure 6.1.

Figure 6 1 **Distribution of countries in the Americas across four different scenarios of human development (as measured by the human development index – HDI, which accounts for education, health and income) and environmental performance (as measured by the environmental performance index – EPI, which accounts for governance related to protection of ecosystems and protection of human health as related to water, air and disaster risks).**

Countries in the green quadrant are those that perform well on both fronts (USA, Canada, Argentina, Chile, Uruguay, Panama, Trinidad and Tobago, Cuba, Costa Rica, Venezuela, Mexico, Brazil, Peru, Dominica, Colombia, Jamaica, Belize, Dominican Republic). In the grey quadrant, we find countries that have a good environmental performance, but have medium to low HDI (Paraguay, Bolivia, Guyana). In the purple quadrant, the opposite situation: high HDI, but medium to low EPI (Bahamas, Ecuador, Suriname, Antigua and Barbuda, Barbados). Finally, in the red quadrant there are countries that perform poorly both in terms of human development and environmental governance (Guatemala, Honduras, El Salvador, Nicaragua, Haiti). The correlation is significant at $p<0.05$. Source: Hsu *et al.*(2016) and UNDP (2015).



All across the Americas there has been steady economic growth, accelerated urbanization, and significant demographic changes in the last decade, despite unevenness and inequality (Chapter 4). There is evidence that poverty and inequality are decreasing at a slow pace in the region, particularly in Latin America (ECLAC, 2011; UN, 2015a; Chapter 4), which remains as one of the most unequal regions in the world (ECLAC, 2016). Steffen et al. (2015a) defined the process of development that took place from 2000 to 2010 as an extension of the ´Great Acceleration´ described for the period from 1750 to 2000. This has boosted patterns of production and consumption in the region, which in turn has also been a key driver of environmental impact (Visconti et al., 2015).

Climate change also remains high on the agenda within the region (UN, 2016). Despite a high environmental performance index, the USA in 2013 accounted for the second highest share (16%) of global greenhouse gas emission, and Brazil, Mexico and Canada were among the top 15 emitters worldwide (Boden et al., 2015). Furthermore, although poverty alleviation and development initiatives have improved adaptive capacity across the region (McGray et al., 2007; Magrin et al., 2014), dramatic forecasts of biome shifts due to global climate change on regional biomes where many people live (e.g. North American prairies, tropical rainforests, tropical alpine environments, the Brazilian caatinga, and coastal/marine environments such as in the Caribbean) indicate that there are limits for adaptation (Seddon et al., 2016). Climate change and biosphere integrity (including massive land conversion, soil and air pollution, and ocean acidification) – although contested by some authors (e.g. Brook et al., 2013), have been recognized as planetary boundaries mankind persistently trespasses (Steffen et al., 2015b), and this is a serious concern for the countries of the Americas.

Countries within the Americas continue to face the challenge of decoupling economic growth and resource consumption. The main challenges and opportunities for protecting and sustainably using the region's biodiversity and ecosystem services are intricately tied to the region's distinct contrasts. On the one hand, the Americas hold what is probably the largest wealth of renewable natural resources on the planet (Mittermeier et al., 2002; 2005), along with a number of creative policies and governance mechanisms to protect and use natural wealth sustainably (Chapters 2, 3). On the other hand, the Americas is also the region with the largest area of agricultural expansion in recent years (Foley et al., 2011), an increasing potential for exploitation of extractives (CEPAL, 2012), and the highest proportion of urban population on the planet (Chapter 4; World Bank, 2012; Magrin et al., 2014).

Another outcome of these combined features is a landscape mosaic of socioecological systems that often imply distinct governance arrangements. For instance, conservation incentives such as the schemes broadly known as payment for ecosystem services (PES) are now more common in Central and South America than anywhere else in the world (Balvanera et al., 2012; Magrin et al., 2014). In terms of percentage of land protected, North America, Mesoamerica, Caribbean and South America hold some of the highest values in the world (Chape et al., 2005). Brazil alone was responsible for 70% of new land brought under protection across the globe between 2003 and 2008 (Jenkins & Joppa, 2009), although most of that was concentrated on the Amazon biome. Despite pockets of noteworthy progress, many policies that define and guide conservation incentives are designed almost solely from an environmental standpoint. In parallel, the outcome of business-as-usual development policies that do not account for the socio-ecological component has often been widespread, unsustainable land use change in rural and urban areas that eventually drive climate change (Chapter 4; Magrin et al., 2014; Nurse et al., 2014; Romero-Lankao et al., 2014).

Reconciling nature conservation and socio-economic development, especially in the context of the 2030 Sustainability Agenda, is the main challenge for the Americas (CEPAL, 2015); and achieving more sustainable use of biological resources is crucial for societies both in and outside of the region, especially in the context of a changing climate (Lucas et al., 2014). From a regional perspective, the clustering of countries at the center of Figure 6.1 suggests some level of similarity in the socio-ecological challenges faced by countries. And although each country will need to tailor development strategies and pathways to suit its own context, regional and subregional cooperation could enhance the exchange of solutions (Ölund-Wingqvist, 2009), and this could potentially accelerate the region's progress towards meeting the Sustainable Development Goals (SDGs).

6.1.2 Our approach to assessing governance and policy

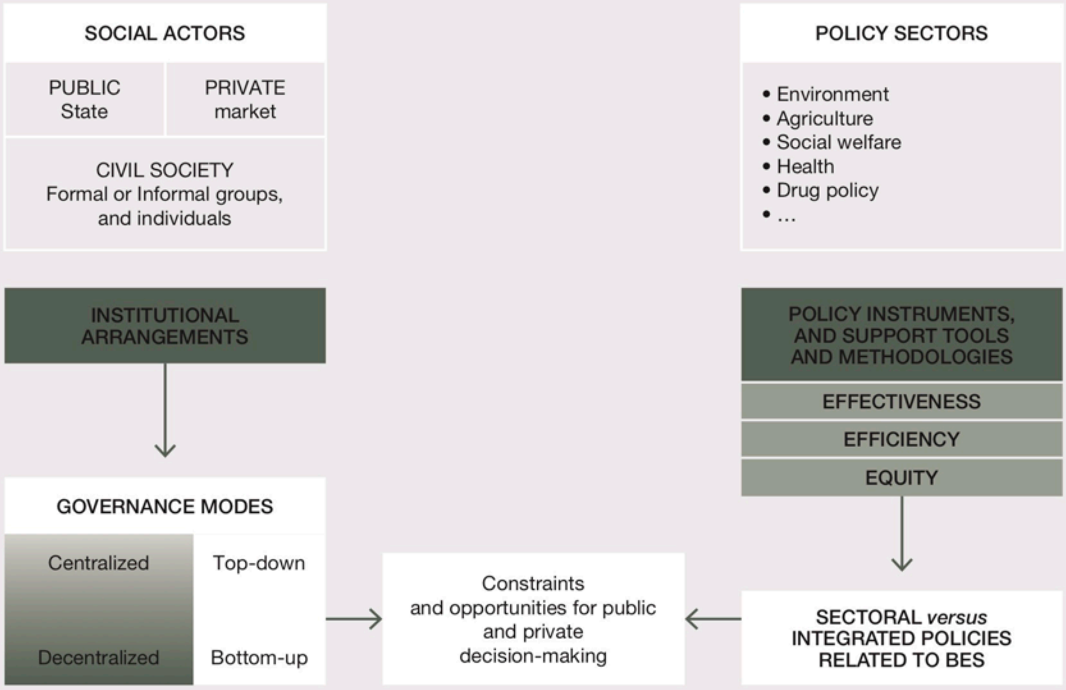
This Chapter starts from the premise that biodiversity and ecosystem services is an important consideration in the sustainability transition process, whereby a given society moves away from unsustainable development trajectories towards a sustainable development paradigm. Such transition processes entail a great deal of complexity, are pushed forward by policies, and are supported by governance options that vary in impact and success (Figure 6.2).

This Chapter follows a set of principles to select the main policies and their respective components that will receive attention in the following sections: (1) To highlight the relevance of the selected options of governance systems and policies for the conservation and sustainable use of biodiversity and ecosystem services, long-term human well-being and sustainable development, rather than to prescribe specific policies or actions. In the process of examining institutions (rules-in-use) and institutional arrangements (formal or informal regimes and coalitions for collective action) in place in the Americas at different levels, equal weight is given to options described as solutions or apparent successes as compared to options described as problematic or challenging. (2) To examine the entire cycle of selected governance systems whenever possible: agenda setting, design, implementation, monitoring and evaluation. (3) To assess feasibility of implementation, scalability from local to higher governance levels, sustainability in time, types of governance systems and processes, institutional capacity and resource allocation, relationship to the private sector, and policy integration for all selected policies. Whenever applicable, the Chapter will take account of potential leakage and spillover effects amongst territories and sectors, be it at regional, subregional, national or sub-national levels. (4) To provide a balanced view among the four macro subregions defined for the Americas: North America (Canada and the USA), Mesoamerica (Mexico and Central America), Caribbean, and South America.

Unlike the previous Chapters in this assessment, treatment is not given to specific biomes - unless in relevant cases where biome-specific policies exist. Rather, a fair balance between inland and marine cases is provided. The Chapter will also attempt to balance the relative effectiveness of policies in cases where there is socio-economic stratification.

Figure 6 2 **Approach to governance and policy related to biodiversity and ecosystem services.**

Left: the interplay between social actors, via institutional arrangements, characterizes the governance modes, which can be more or less centralized depending on the balance between top-down and bottom-up processes and of power relations between actors. Right: societal sectors are characterized by a diversity of sectoral policy instruments, support tools and methodologies. These are more or less influenced by the principles of effectiveness, efficiency and equity, which will determine the level of integration of policies. Governance arrangements (left) and policies (right) interact and result in constraints and opportunities for public and private decision-making. Source: own representation inspired in an analogous scheme produced by the Europe and Central Asia Regional Assessment.



6.2 Sectoral versus integrated policies

Policies are often designed from a sectoral perspective (Gomar, 2014; Lima et al., 2017). The topics of interest to IPBES - biodiversity, ecosystem services, and human wellbeing - comprise sectors themselves: environmental sector (in the case of biodiversity and ecosystem services) and social welfare sector (at least partly, for the case of human wellbeing), which tend to have specific policies. Conventional development and market forces also display sectoral policies that affect and are affected by biodiversity and ecosystem services and human wellbeing. There is an increasing body of evidence showing that environmental conflicts and unsustainability emerge largely from the lack of integration between/ among sectors and between/among sectoral policies, particularly when development and market policies do not account for environmental and/or social issues (e.g. Franks et al., 2014) and vice-versa (e.g. Adams & Hutton, 2007). This is especially important given the finding in Chapter 4 that impacts on biodiversity and ecosystem services are often the result of multiple driving forces working in tandem.

Bennett et al. (2015) have stated that focus on a single driver of change is often the result of a problem-centered, rather than a community-centered, approach. Based on an extensive review, these authors argue that various socioeconomic and biophysical changes may take place concurrently and at multiple scales to result in different outcomes for communities in different places. This is why, they propose, that when the predominant focus of vulnerability and adaptation research, policy and practice lie solely on one problem (e.g. biodiversity conservation or climate change) it undermines the complexity of multiple interacting variables. A community-centered approach requires multifactorial, transdisciplinary analysis and subsequent interventions, but remains more theoretical than empirical.

One example that relates to the argument of Bennett et al. (2015) is the impact of policies to combat narcotraffic on policies to combat deforestation. For instance, policies to eradicate drug plantations (coca, opium poppy and marijuana) in the Andes often push growers into ecologically sensitive zones causing environmental impacts in those areas (McSweeney et al., 2014). In addition to this case study, these authors use examples from Mexico, Honduras, Guatemala to postulate that well-targeted narcopolicy reforms could yield important socio-ecological benefits, by reducing pressure on forests and local communities (including indigenous ones) while reinforcing governance of protected areas. A second example is related to Reducing Emissions from Deforestation and Forest Degradation Plus (REDD+ (the “+” refers to additional benefits, such as those derived from biodiversity conservation, for instance)). Phelps et al. (2012) illustrate the potentially challenging trade-offs between climate change and the “+” related to biodiversity conservation; and the demands and needs related to livelihoods of forest-dependent communities, extractive industries and national economies (see 6.4.2 for specific examples in the region). Moreover, Faith (2014) warns that over focus on local carbon/biodiversity win-wins could mean a collapse in the regional capacity to conserve biodiversity.

Greater mainstreaming of biodiversity and ecosystem services considerations into important development sectors such as energy and agriculture is occurring in many governments, but scope for substantially more progress has been identified (CBD, 2016a). In the region and elsewhere, there is a reported recent trend toward developing more domestic energy sources partly due to political uncertainties in the relationships with some oil-rich nations and part due to the desire to maintain energy security (Jones et al., 2015). This trend is exemplified by the USA, where wind energy increased 23-fold since 2000, and natural gas production has risen by almost 21% over the last two decades. While changes in energy systems seems to be taking place fast, scientific literature on the relationship between the energy sector and biodiversity and ecosystem services remains biased geographically, with the USA and Canada housing most of the studies (Jones et al., 2015). Political decision-making in this regard is therefore uneven in most countries. Two facts, however, are relevant for policy design and implementation in the energy sector during its transition to a model that is less carbon intensive, and less harmful to biodiversity and ecosystem services. First, there are no renewable energy pathways that have zero environmental impact, especially if they are to be deployed at a large-scale (Gasparatos et al., 2017). Thus, compensation and offset logic ought to be applied to energy projects (see 6.4.2.2) and biodiversity and ecosystem services policy instruments should also be used in energy policy design. Central America has been pointed out as a particularly vulnerable region in terms of biodiversity impacts from renewable energy expansion (Santangeli et al., 2016). Secondly, most energy sources depend on good flow of ecosystem services. Typical examples are water for hydropower (see Medeiros et al., 2011, on the role of protected areas in Brazil; and Sáenz et al., 2014, on the role of cloud forests in Colombia) and pollinators for agriculture (in the region, the agricultural Gross Domestic Product (GDP) of countries like USA, Brazil and Argentina are largely dependent on pollinators; Lautenbach et al., 2012; see also Chapter 2).

There are also persistent challenges to mainstreaming biodiversity and ecosystem services into the agricultural sector. Most policies and practices are designed specifically either for farm, or for landscape, or for the agri-food system that has national to global reach (Foley et al., 2011). Moreover, agri-food systems are under pressure from external factors - such as globalisation, climate change, and scarcity of resources - and internal factors - such as changes in market relations, asymmetric price transmission, input suppliers and retails concentration, changing consumer demands (Hubeau et al., 2017; Lowitt et al., 2015). In turn, such global systems impact biodiversity and ecosystem services (Moran & Kanemoto, 2017). For instance, in the case of Brazil, the third largest agricultural exporter in the world (after EU and USA; Handford et al., 2015), recent national-level legislation on conservation and restoration within private properties to be delivered at farm level will have positive landscape consequences related to connectivity and protection of biodiversity and ecosystem services, and will feed into the country´s commitments at various global conventions (Soares-Filho et al., 2014; Brancalion et al., 2016; Scarano, 2017; see also 6.6.3). In parallel, governmental incentives for low carbon agriculture (Soares-Filho et al., 2014; Rodríguez-Osuña et al., in press) and existing state-level payment for ecosystem services legislation are well aligned with that

(Zanella et al., 2014). In contrast, lack of policy integration is perceived when existing farm-level policies and standards (see Handford et al., 2015) do not keep the country from being the largest worldwide user of agrochemicals in commodities (Gerage et al., 2017); when logistic and infrastructural limitations at national level cause large global greenhouse gas emissions from transportation of agricultural goods and beef (Soysal et al., 2014); when export policies of commodities create biodiversity footprint hotspots driven by market demands of the USA and the European Union (Moran & Kanemoto, 2017); and when many smallholder farmers who produce food remain poor and vulnerable to climate change (Burney et al., 2014; Guedes et al., 2014). These contradictory policies related to Brazilian agriculture and its impacts on biodiversity and ecosystem services are being transferred to Mozambique to some extent, in the realm of the cooperation of the two countries in this field (Zanella & Milhorance, 2015).

The role and impact of private sector in biodiversity and ecosystem services management is another key dimension. Private sector practices are often influenced by government-led policies and vice-versa. Franks et al. (2014) showed major losses of mining and hydrocarbon companies due to conflicts that emerged locally when environmental (e.g. biodiversity and ecosystem services) and social (e.g. consent, culture) variables were not accounted for in their policies and governance systems, including countries such as Argentina and Peru. They concluded that to ensure sound management of environmental and social risks and to deal constructively with conflicts, the policy environment should encourage: (1) effective predictive assessment and management of environmental and social impacts; (2) greater community involvement in dialogue and decision-making during the early stages of projects (including addressing community held expectations for consent); (3) the formalization of such dialogue into agreements between companies and their employees, indigenous peoples, and communities; and (4) the implementation of conflict resolution and grievance handling approaches. This rationale is in harmony with the assessment made by Jaskoski (2014) who, in a case study in Peru, found that reduced space for community participation in the environmental impact assessment process led to the stalling of major extractive projects.

6.3 Governance

6.3.1 Moving from a state-centered approach to greater participation

Diverse forms of socio-ecological governance strategies are being practiced worldwide, and it is now clear that market, state, or civil society-based strategies depend on support from other domains of social interactions for their efficacy. The complex nature of socioecological challenges and the occasional reluctance or inability of nation states to regulate the sources of these problems or to enforce solutions lead to increasingly decentralized governance schemes, where nonstate actors are capable of generating innovative solutions (Lemos & Agrawal, 2006). This is an emerging trend in the Americas.

Until the 1990’s, state-centered governance dominated most of the Americas region - particularly Latin America, where several countries were under military dictatorship -, with top-down decision-making procedures controlled by a technocratic elite and grounded in a nationalist discourse of state sovereignty (Castro et al., 2016). In the 1990’s, most Latin American societies - which were not already democratic - went through a process of democratization but, at the same time, continued to be influenced by policies from international institutions, particularly the International Monetary Fund, the World Bank and the Inter-American Development Bank (Liverman & Vilas, 2006). These policies called for a market-based approach to self-governance, through self-designed corporate mechanisms such as social responsibility, certification and compensation schemes (Castro et al., 2016).

At the same time, another type of self-governance approach began to become visible: governance systems relying primarily on collective action to regulate the access to and use of natural resources (common-pool resources, or simply the commons). Stemming from evidence collected in multiple disciplines, this mode of self-governance gained the attention of society by environmental justice movements and transnational activism networks (Castro et al., 2016). Defying a widespread theory that resource users are incapable of self-organizing to maintain their resources – and therefore the only way to avoid a “tragedy of the commons” (sensu Hardin, 1968) would be privatization or State rigorous control –, an extensive body of literature shows many cases where human groups (with an important presence of indigenous peoples) have used their resources sustainably for generations (e.g. Brondizio et al., 2009; Gibson et al., 2000; Ostrom et al., 2002), whereas some government policies have accelerated resource depletion (Ostrom, 2009).

Analyses of cases all around the world support the idea that groups who are capable of self-organizing to successfully manage their resources tend to follow some principles such as clearly defined boundaries, equitable rules for sharing benefits and costs, effective monitoring arrangements, graduated sanctions for those who violate rules, mechanisms for conflict resolution, and recognition of rights to organize (Cox et al., 2010; Ostrom, 1990). Sattler et al. (2016) reached similar conclusions in an analysis of four cases of multilevel community governance in Latin America (three in Brazil, one in Costa Rica). Furthermore, they acknowledged that complex solutions that work for a specific context often cannot be transferred directly to another context.

Participatory governance emerged in this context in the 2000s and became a central element of environmental governance in the Americas (Castro et al., 2016). Viewed as an alternative capable of deepening democracy and citizenship, it seeks to integrate environment with other societal concerns – such as poverty alleviation, inclusion of minorities and local and indigenous populations, and social justice; and to devise strategies and solutions that differ from the top-down ones, in terms of greater inclusion of local knowledge, greater learning capacities, and improved accountability (Fung & Wright, 2001). In state-, community- or market-based governance, participatory governance is based on partnerships between relevant actors to set goals and to design and implement initiatives. It ranges from models of partnership between state and local communities developing a plan for territories, to more complex arrangements including multistakeholders and multiscale institutions. It represents a new layer in hybrid governance models composed by state-centered, market-based and local-based mechanisms (Castro et al., 2016) (see also Table 6.1 for more examples across the region).

Participatory governance has not been without its challenges however. The effectiveness of these arrangements depends on the manner in which different worldviews and interests are negotiated, how problems are prioritized, and how compatible the proposed solutions are with the social, institutional and environmental context. Examples of success stories range from the soybean moratorium in Brazilian Amazonia to watershed management in Montana, USA. Soybean moratorium was encouraged by non-governmental organizations (NGOs) and soybean retailers and resulted in significant reduction in deforestation due to soybean production (Nepstad et al., 2014; see also 6.3.4). In Montana´s Yellowstone River Basin, dissensus – as a particular aspect of collaboration in a collaborative planning effort for water use measurement– has been essential to disrupt modes of inquiry, open alternative perspectives, and provide innovative possibilities, even among sanctioned participant voices operating within otherwise established, depoliticized governing arenas (Anderson et al., 2016).

Therefore, a diversity of cases across policy areas, levels of economic development, and political cultures around the globe suggest that partnerships and participatory deliberative processes contribute to a large class of problem-solving situations and can support successful governance of socio-ecological systems (Fung & Wright, 2001; Tucker, 2010; see also 6.4.2, 6.4.3). However, there is recent theory on the existence of conditioning factors for participation to lead to successful environmental outcomes (e.g. Newig et al., 2017), which still requires testing. One potential weakness in participatory processes occurs when participation is treated or included as a façade. In such cases, state and/or other privileged actors retain the authority to govern within a given arena - while giving the impression of being more decentralized or democratic - while less privileged stakeholders merely approve top-down designed policies (Anderson et al., 2016).

Table 6.1Examples of participatory processes in the Americas that take place in protected areas and/or in community-based management areas: public policies with the involvement and participation of social actors in the design, implementation, monitoring and/or evaluation stages

Instrument	Type	Country (and sub-national unit)	Brief description
Protected areas	Shared management with NGOs, companies, and/or other civil society institutions	Brazil (e.g. Amazonas and São Paulo states)	Often informal support to management by non-governmental institutions (Koury & Guimarães, 2012; Borrini-Feyerabend *et al*., 2004; Lima & Pozzobon, 2005; Maccord *et al*., 2007)
		Canada	Organizations with indigenous representation in co-management agreements (Armitage, 2005; Borrini-Feyerabend *et al*., 2004)
		Paraguay	Co-management with participation of the private sector (Sienra & Medina, 2013)
		St. Lucia, Trinidad and Tobago,	Co-management with participation of the private sector and other groups of interest (Adger *et al.,* 2006)
	Shared management with local communities	Argentina (Patagonia)	Involvement of indigenous and local coastal communities (Garcia, 2003)
		Brazil	Involvement and participation of local communities in the decision-making process through deliberative or advisory councils (Queiroz, 2005; Queiroz & Peralta, 2006; Silvano *et al*., 2014)
		Canadá	Involvement of indigenous communities in territorial management (Armitage, 2005; Borrini-Feyerabend *et al*., 2004)
		Mexico	Participatory land management by local communities (Porter-Bolland *et al*., 2013)
		Paraguay	Involvement of indigenous populations on management of natural áreas (Fogel, 2007)
	Control and surveillance	Brazil	Voluntary agents from local communities (Souza & Queiroz, 2008; Lima & Pozzobon, 2005)
	Private PAs officially recognized	Belize, Brazil, Costa Rica, Ecuador, Dominican Republic, Guatemala, Peru, Trinidad and Tobago, Venezuela	Mostly dedicated to ecotourism (Langholz, 1996)
Community-based management	Fisheries	Antigua, Argentina, Barbados, Belize, Chile, Colombia, Costa Rica, Cuba, Dominica, Dominican Republic, Ecuador, Guatemala, Guyana, Mexico, Nicaragua, Panama, Peru, Puerto Rico, St. Vincent & Granadines, Trinidad and Tobago, Uruguay, Venezuela	Participatory management by coastal communities and private sector: marine (Salas *et al*., 2007)
		Brazil	Participatory management by riverine communities in Amazonia: freshwater (Castello *et al*., 2009, 2011; Kalikoski *et al*., 2009; McGrath *et al*., 1993; Silvano *et al*., 2014; Arantes & Freitas, 2016)
			Participatory management by coastal communities: marine (Diegues, 2008; Freitas & Tagliani, 2009; Lopes *et al*., 2013; Reis & D´Incao, 2000; Schafer & Reis, 2008; Salas *et al*., 2007)
		Canada	Participatory management by coastal communities: marine (Kearney *et al.,* 2007; Pinkerton & Weinstein, 1995; Wiber *et al*., 2004)
		Chile, Mexico	Participatory management of small scale benthonic fisheries (Basurto *et al.,* 2013)
		Mexico	Participatory management of watersheds, water and fisheries (Porter-Bolland *et al*., 2013)
		St. Lucia, Trinidad and Tobago	Participatory management of coastal fisheries in National Parks (Adger *et al*., 2006)
		USA (Alaska and Washington)	Participatory management of freshwater salmon by the Pacific (Kellert *et al*., 2000)
	Forestry	Belize, Costa Rica, El Salvador, Guatemala, Panamá	Forest community management (Gómez & Méndez, 2005; Hogdon *et al*., 2015; Larson, 2003; 2005; Primack *et al*., 1998; Radachowsky *et al.,* 2011; Sayer & Campbell, 2004)

Community-based management (cont.)		Bolivia, Colombia, Peru	Community management associated to strengthening of family-based agriculture in the Andes (Sayer & Campbell, 2004; Larson, 2003; 2005)
		Brazil	Low impact, timber and non-timber management, in Amazonian flooded forests (Schöngart & Queiroz, 2010; Larson, 2003)
		Canada	Management by indigenous peoples and their organizations (Natcher & Hickey, 2002)
		Honduras, Nicaragua	Participatory management with communities, organizations, local and central government (Nygren, 2005; Larson, 2003; 2005; Sayer&Campbell, 2004)
		Mexico	Management inside and outside protected areas, by community-based companies (Ellis & Porter-Bolland, 2008; Porter-Bolland *et al*., 2013)
		US	Collaborative management and monitoring by community-based organizations (Fernandez-Gimenez *et al*., 2008; Meffe *et al*., 2002)
	Hunting	Canada	Participatory management of hunting and fishing by indigenous communities (Armitage, 2005)
		United States	Community management of hunting (Decker *et al*., 2004; Meffe *et al*., 2002)
	Water	Brazil	Multistakeholder watershed management and governance of water resources
		Paraguay	Management by indigenous peoples inside their territories (Fogel, 2007)
		United States	Participatory co-management of watersheds and water resources (Rhoads *et al*., 1999)
Monitoring	Biodiversity and natural resource monitoring	Bolivia, Canada, Colombia, Ecuador, México, Nicaragua, Peru	Citizen monitoring by using transects, camera traps, species lists (Danielsen *et al*., 2005; 2008; 2014; Pasteur & Blauert, 2000)
	Monitoring of invasions of protected areas	Ecuador	Community-based, with no participation of official agencies (Danielsen *et al*., 2008)

6.3.2 Addressing socioecological complexity in governance systems

Facing the intrinsic complexity of coupled human-environment systems and, consequently, of contemporary problems, societies increasingly acknowledge the absence of one-size-fits-all solutions; in other words, there are no panaceas for socio-ecological governance (Ostrom et al., 2007). Complexity here means that these systems are self-organizing, interconnected within and across scales and levels, and their trajectories are highly unpredictable, nonlinear, and frequently surprising. Therefore, some argue that in order to manage complex problems, governance approaches should aim to build socio-ecological resilience, as a perspective for understanding how co-evolving societies and natural systems can cope with, and develop from, disturbances and change (Duit et al., 2010; Walker & Salt, 2012). A resilience approach to governance may enable understanding of the dynamics of rapid, interlinked and multiscale change, as decision-makers try to deal with converging trends of global interconnectedness and increasing pressure on socio-ecological systems. However, criticism on resilience thinking ranges from lack of consensus around the definition of resilience to lack of clarity or difficulties at establishing its practical application (see Walsh-Dilley et al., 2016). Walker and Salt (2012) argue that resilience practice is largely dependent on understanding limits, thresholds, tipping points, so as to have a perspective of regime shifts – which are often difficult to detect.

Biggs et al. (2012) present a set of general principles for building resilience into socio-ecological systems, which are discussed specifically in terms of enhancing the resilience of ecosystem services. The seven principles are (1) maintain diversity and redundancy, (2) manage connectivity, (3) manage slow variables (e.g. composition of soil or sediment nutrient) and feedbacks (i.e., slow responses in the system to change in given variables), (4) foster an understanding of social-ecological systems as complex adaptive systems, (5) encourage learning and experimentation, (6) broaden participation, and (7) promote polycentric governance systems. In accordance with this view, there is rich discussion in the literature, based both on empirical data and theoretical construction, proposing adaptive management, adaptive co-management and adaptive governance as systems more suitable to overcome contemporary socio-environmental problems. Adaptive management emphasizes learning and uses structured experimentation in combination with flexibility to foster learning. Adaptive co-management explicitly links learning and collaboration to facilitate effective governance. Adaptive governance connects individuals, organizations, agencies, and institutions at multiple organizational levels. Adaptive governance systems often self-organize as social networks with teams and actor groups that form a learning environment to draw on various knowledge systems and experiences to tackle complex environmental issues (Stockholm University, 2014). Knowledge generation, bridging organizations, social learning and collaboration are in the core of these systems (Armitage et al., 2009; Berkes, 2009; Folke et al., 2005).

Similarly, Crozier (2008) points out that generating knowledge, authority and legitimacy to effectively respond to societal issues requires the involvement of multiple actors from different scales and levels through interactive structures and processes to stimulate communication and the sharing of responsibilities among actors. Networks seem to offer a way to manage processes that involve multiple actors with diverse interests and orientations. Different evaluations of network governance agree on the importance of facilitating interactive processes, mediating interactions between actors, and focusing on goal searching rather than goal setting (Crozier, 2008; Scarlett & McKinney, 2016). In the region, there are a number of examples of network governance to achieve goals related to restoration (e.g. Americas Longleaf Restoration Initiative - US, Scarlett and McKinney, 2016; Atlantic Forest Restoration Pact – Brazil, Pinto et al., 2014), control of invasive alien species (e.g. Invasive Spartina Project – US, Lubell et al., 2017), fisheries (e.g. Special Fisheries Conservation Areas – Jamaica, Alexander et al., 2016), among others. The Fire Learning Network, in the USA, exemplifies a network based on an interacting process of adaptive learning. It facilitates information flow across scales, involving diverse actors, stimulating innovative solutions, influencing plans and policies, and then using this learning to enable further experimentation and innovation. It builds socio-ecological resilience by overcoming the rigidity traps that characterize many natural resource management bureaucracies (Butler & Goldstein, 2010; see also http://fireadaptednetwork.org; and https://www.conservationgateway.org/ConservationPractices/FireLandscapes/FireLearningNetwork/Pages/fire-learning-network.aspx)

6.3.3 Achieving better integration in policy through effective governance

Design and implementation of multidimensional, multifactoral policies require effective governance systems. For instance, it is estimated that the developing world suffers 140,000 child deaths and loses $1 trillion every year because of corruption and poor governance, which is a monetary measure of the negative costs of ineffective governance (Joshi et al., 2015). Corruption is still a major issue in Latin America and the Caribbean (CEPAL, 2007; Kaufmann, 2015), despite efforts to combat these through strengthened national regulations and increased international cooperation (OECD, 2016). Joshi et al. (2015) showed that high-income countries (including USA and Canada) had, by 2010, the highest composite governance (0.86) and HDI = 0.87. Latin America and the Caribbean came second in this world ranking, both in terms of composite governance index (0.66) and HDI (0.70). These data are in harmony with Figure 6.1 and suggest that some Latin America and the Caribbean countries still have room for improvement in terms of governance effectiveness, which might be an obstacle for design and implementation of integrated policies that foster sustainability.

However, the Latin America and the Caribbean subregion displays some interesting examples of effective governance systems related to cross-cutting issues and policies. Estrada-Carmona et al. (2014) surveyed 104 initiatives of integrated landscape management in 21 Latin America and the Caribbean countries, which aim at reconciling food production, livelihood improvement and biodiversity and ecosystem services conservation. They found that positive results were often related to institutional planning and coordination of the local governance systems, whereas setbacks and challenges were related to the long time span necessary to bring results to scale, unsupportive public policy frameworks, and lack of private sector engagement. Non-governmental organizations were important stakeholders in 87% of the initiatives surveyed.

6.3.4 Factoring scale into governance arrangements

Multiple socioeconomic and biophysical changes take place simultaneously at different scales and levels, interacting to produce different outcomes for communities in different places (Bennett et al., 2015). There is often no fixed scale or level that is sufficiently appropriate for governing ecosystems and the services they provide (Brondizio et al., 2009). Sustainability in agriculture will require good practices at all three levels – farm, landscape and market: although policies are usually set at national or sub-national level, they normally respond to market demands (which can range from local to international) and are implemented at farm and landscape levels. However, it is only rarely that all three levels are dealt with in an integrated fashion, scientifically or politically (see Clapp, 2015).

Thus, socio-ecological challenges often have a multilevel nature and require connecting different institutions across levels to facilitate governance and build on social capital that is essential for the long-term protection of ecosystems and the well-being of human populations (Brondizio et al., 2009). Cross-scale and cross-level problems may emerge when this is not considered. Leakage is a typical case. For instance, Lui and Coomes (2015) showed that for c. 80% of 60 protected areas (including 20 in tropical America) deforestation rates increased gradually from their interiors to the outer periphery of their buffer zones. Another example is that of the soybean moratorium in Brazil: an arrangement whereby major soybean traders agreed not to purchase soy grown on lands deforested after July 2006. The result of this movement, incentivized by NGOs and soybean retailers, was that deforestation in Brazilian Amazon due to soy expansion dropped to less than 1% (Rudorff et al., 2011; Gibbs et al., 2015). However, deforestation for soy expansion leaked into neighbouring biomes such as the Brazilian Cerrado (Morton et al., 2016).

Environmental problems and the human actions to overcome them frequently display a scale mismatch. A common mismatch on time-scale arises, for example, when public policies depend on short electoral cycles that conflict with long-term planning needs (Cash et al., 2006). Thus, integration across functions, space, time, institutions, fields of knowledge, governance, and other dimensions are important and essential for the sustainability of ecosystems and societies (Ascher, 2007). Regarding governance scale, policies can be local, subnational, national, regional or global. How lessons learnt on governance scale up from local practice to become policies at any level, and how policies agreed upon and designed globally or nationally are mainstreamed into local practices is a matter of interest to allow amplification of solutions and best practices.

One important concept regarding cross-scale governance, and top-down/bottom-up relationships, is that of boundary objects. Marine protected areas have been defined by Gray et al. (2014) as one such boundary objects, since they "range in size, purpose, resource use policies, and governance structures, for example, from large no-take areas identified for their ecological value and administered by states, to small, multi-use areas protected by communities". Thus, while individual marine protected areas are the outcome of particular local-to-national political processes, the cumulative global increase in these areas´ number and coverage is the result of a coordinated international effort.

In insular Caribbean, for instance, a combination of national initiatives, with regional efforts such as the Caribbean Challenge Initiative to protect by 2020 "at least 20% of nearshore marine and coastal habitats" and international efforts of multilaterals and NGOs to ensure data consistency, have resulted in an marine protected areas coverage comparable to global figures and not as far below global CBD (United Nations Convention of Biological Diversity) targets (Knowles et al., 2016). For an example in another realm, Holden (2013) suggests that indicator systems should be applied as boundary objects, in other words, tools which "open up dialogue, information sharing, learning and consensus-building across different policy boundaries: between experts and nonexperts, formal government and different nongovernment actors, higher-order governments and lower-order governments". The application of this approach in the urban context in Seattle (USA) and Vancouver (Canada), according to this study, indicates the usability of non-governmental indicator systems designed for use as boundary objects, as a leap forward for indicator work aiming to change policy, from a governance perspective.

6.3.5 Indigenous and local knowledge systems
Especially considering the issue of scale, and in the context of the Americas, it is particularly relevant to address local and indigenous groups that have their own governance or environmental management systems based on an extensive and detailed ecological knowledge accumulated throughout several generations. The region has hundreds of indigenous communities as well as other local communities that have a close and traditional dependence on biological resources. For instance, while in most countries indigenous peoples are perceived as minorities, in some (e.g. Bolivia, Guatemala, Mexico, Peru) they constitute a significantly large percentage of the population. Other local groups living in traditional dependence on biological resources include afro-rural communities (e.g. Brazil, Ecuador, Panama, Surinam – IPEA, 2012), raizales in Caribbean Colombia, and caiçaras in coastal Brazil, among others. Many but not all governments[32] acknowledge the ethnic and cultural identity of these populations' governance and/or management systems and the rights to coexist with the mainstream (western-based) governance system. However, both in countries with and without acknowledgement of such indigenous and local governance systems, there are accounts of many related conflicts both with governments and with the private sector (e.g. Franks et al., 2014; Haslam & Tanimoune, 2016; see also 6.4.1.1).

Many such systems related to local and indigenous groups are based on worldviews that consider biotic, abiotic and human dimensions as integral parts of a whole. Although several countries in the Americas formally recognize such self-governance systems, in most cases these groups are marginalized and have little political power under the authority of a central government (Vinding & Jensen, 2016). In a few cases,

[32] Countries in the Americas that ratified the Indigenous and Tribal Peoples Convention n.169, from the International Labour Organization (1989): Argentina, Bolivia, Brazil, Chile, Colombia, Costa Rica, Dominica, Ecuador, Guatemala, Honduras, Mexico, Nicaragua, Paraguay, Peru, Venezuela. Source: http://www.ilo.org/dyn/normlex/en/f?p=NORMLEXPUB:11300:0::NO::P11300_INSTRUMENT

such as the Plurinational State of Bolivia (Pacheco, 2014; UNEP, 2013) and aboriginal peoples (including The First Nations) in Canada (Slowley, 2001; Preston, 2016), there is enough political decentralization allowing for the coexistence of self-governing and western-based systems – although conflicts occasionally occur. In Tomave, Bolivian Andes, the Ayllu Sullka people have their own political and social system, an autonomous government that sometimes share decisions with the State authorities, but, in general, decisions are taken from the bottom-up, by consensus of assemblies. With the exception of the "wise elder" of the communities, there is rotation in every other government's position. The Ayllu Sullka ecological-territorial management is based on the concept of living well, in harmony and balance with Mother Nature, and depends on principles such as: indigenous government; exchange of products and seeds; integral and communal management of the territory; food sovereignty; spiritual practices in sacred locations and medicinal plants; communal land ownership, including land redistribution to accommodate the needs of all families. This system ensures the conservation of the ecosystem as a whole, including cultivated plants, especially potatoes and quinua, and domesticated animals, especially camelids (Mamani Machaca, 2017).

In Tungurahua, Ecuador, local communities challenged an international model of watershed management reform that coupled conservation with markets for ecosystem services, and negotiated with transnational advocates to create an alternative model rooted in indigenous norms (Kauffman & Martin, 2014). They did not reject the idea of reforming watershed management, but aimed to do so by realizing the Quichua concept "sumah hawsay" (*buen vivir* in Spanish or wellbeing in English), which refers to living in harmony with nature, rather than dominating nature or removing human presence through preservation. The government of Ecuador has brought the attention of this case to various international fora. It shows how new environmental governance regimes can emerge locally by a participatory process where global, international agendas are critically questioned and revised according to local culture and principles, and how, in turn, such learning poses a reflection at global level that might challenge dominant international norms (see also 6.4.3.2).

The outcomes of the study by Evans et al. (2014) on the perception of REDD by community members in the Amazonian state of Loreto, Peru, have followed a similar logic to the Ecuadorian case (see also Vasseur et al., 2017). Indigenous interviewees were skeptical about REDD's long-term positive impacts for communities and forests, including benefit distribution, and also revealed uncertainty about the future and lack of trust in governance regimes. Community priorities included work opportunities, educational opportunities for their children, and improving the quality of their forest. The author's conclusions were that REDD design should recognize local communities as active participants in global and national climate management. Indeed, a recent study by Ochieng et al. (2016) has shown that whenever overall effectiveness of REDD+ schemes is only moderate it is due to either issues with exercising good governance (e.g. Bolivia) or with lack of ownership of technical methods (e.g. Peru). Further issues regarding carbon benefits from REDD relate to the realism of baselines and the treatment of leakage and permanence (Vitel et al., 2013).

6.4 Policy instruments, support tools and methodologies related to biodiversity and ecosystem services

Several existing policy instruments are applicable to different biodiversity and ecosystem services-related policy types, be they sectoral or integrated. However, they can often be perceived as environment-related only, and being neutral or even negative to socio-economic aspects. This section examines relevant biodiversity and ecosystem services-related instruments and how they relate to human well-being and sustainable development. They are divided into three groups of instruments: regulatory mechanisms, incentive mechanisms, and rights-based approaches. This classification is for schematic purposes only: we understand that there is a significant overlap between these groups. For instance, protected areas and ecosystem restoration are often regulatory, but can emerge out of incentive mechanisms or voluntarily. Their design and implementation can also follow a rights-based approach. Policy instruments are developed and adopted by the use and application of policy support tools and methodologies. The draft guidance of IPBES (IPBES, 2016) defines policy support tools and methodologies as "approaches and techniques based on science and other knowledge systems that can inform and assist policy-making and implementation at local, national, regional and international levels to protect and promote nature, nature's benefits to people, and a good quality of life". These support tools and methodologies have been organized in a typology of families and this section will examine examples from each of them (Box 6.1).

Box 6.1. Families of policy support tools and methodologies

Assembling data and knowledge: this family includes monitoring, indicators, oral history, mapping of ecosystem services, census data, population dynamics.

Assessment and evaluation: this family includes trade-off analysis, management effectiveness, trend analysis, identification and assessment of indigenous and community conserved areas (ICCAs), quantitative modelling, cost-benefit analysis, non-monetary valuation, scenarios.

Participatory processes: this family includes expert interviews, stakeholder consultation, cultural mapping and implications for policy goals and criteria, social media tools.

Selection and design of policy instruments: this family includes instrument impact evaluation, ex-ante evaluation of options and scenarios, designing of individual territory sets or systems of protected areas.

Implementation, outreach and enforcement: this family includes audits, risk-based enforcement efforts, process standards (e.g. ISO), monitoring reporting and verification.

Capacity building: this family includes handbooks, manuals, guides, e-learning resources, webinars, training, education, knowledge sharing.

Social learning, innovation and adaptive governance: this family includes strategic adaptive management and social learning theory.

Source: IPBES Policy support catalogue available at http://ipbes-demo.net/node/140

6.4.1 Regulatory mechanisms

6.4.1.1 Protected areas

Areas of particular importance for biodiversity, ecosystem services and human wellbeing (including protected areas, ICCAs, other area-based conservation measures, and biodiversity, ecological and conservation corridors) are among the main policy instruments that address biodiversity and ecosystem conservation in the region. Supporting tools and methodologies such as species and ecosystem redlists and participatory processes are often used. The region presents a broad diversity in the history of use and application of such instruments and tools.

Protected areas - public, communal and private - have been a key element in biodiversity and ecosystem services conservation, in promoting tourism (see also 6.6.1) and also in generating social and community benefits across the region and elsewhere (Watson et al., 2014). In the Americas, the proportion of protected areas following IUCN (International Union for Conservation of Nature) definition by 2017, was higher than the global average: North America had 11.3% of its terrestrial area protected and 25% of its marine areas protected, Mesoamerica 17.5% terrestrial and 2% marine, Caribbean 17.5% terrestrial and 5.7% marine, and South America 24.0%, and 5.9% marine (UNEP-WCMC and IUCN, 2017). By 2014, Latin America and the Caribbean altogether continued to lead globally with 23% of its land under protection (UN, 2015a). The region has thus been progressing well towards Aichi target 11 (see 6.5.1). Two main questions deriving from this are: how effective are such protected areas (1) for nature conservation and (2) to provide direct and indirect socio-economic development benefits.

On the distribution and coverage of protected areas, there seems to be greater emphasis on forests (especially tropical and subtropical) and other highly diverse ecosystems, at a global level (Anthamatten & Hazen, 2014). This also holds even in terms of other environment-related policies, such as restoration, conservation incentives, etc. (e.g. Overbeck et al., 2015). On the effectiveness of protected areas for biodiversity and ecosystem services conservation, there are mixed viewpoints, but there remains a clear gap regarding impact assessment (Coad et al., 2014; Pressey et al., 2015). While some meta-analyses indicate an overall positive impact of protected areas on conservation (e.g. Bruner et al., 2001; Geldmann et al., 2013), other authors argue that little is known about how much difference protected areas actually make (Pressey et al., 2015). On the positive side, Bruner et al. (2001) analysed 93 parks in 22 tropical countries (34 of them in the Americas) and concluded that most of them are effective, especially at protecting from land clearing and, to a lesser degree, at mitigating logging, hunting, fire, and grazing. As they found the effectiveness of parks to correlate with basic management activities such as enforcement, boundary demarcation, and direct compensation to local communities, they suggest that even modest increases in funding would improve parks effectiveness. This is consistent with the meta-analysis more recently performed by Geldmann et al. (2013) including 35 cases from Central and South America and one from North America, which found a positive impact of protected areas on conservation in 86% of cases. GEFIEO (2015), in a study that analysed 618 projects funded by the Global Environment Facility in protected areas of 137 countries, found that a combination of good governance, effective protected area management, and community engagement explain why protected areas funded by the Global Environment Facility are more effective in delivering conservation outcomes than those not funded.

However, Watson et al. (2014) demonstrated that recent years have seen a decline in the effectiveness of protected areas across the region, with problems such as major budget cuts (e.g. USA), extractive activities inside national parks (e.g. Belize), and increasingly frequent protected area downgrading, downsizing and degazettement (e.g. Brazil). For some species, climate change poses an additional threat to the effectiveness of biodiversity conservation in protected areas in the region, be it at taxa level (e.g. Ferro et al., 2014; Lemes et al., 2013; Loyola et al., 2012; Nori et al., 2015), or from an evolutionary history perspective (Loyola et al., 2014). Geldmann et al. (2013) argue that there is limited evidence for understanding the exact conditions or combinations of circumstances under which this policy instrument succeeds or fails to deliver conservation outcomes.

Marine protected areas are smaller than their terrestrial counterparts in proportional coverage, in the region and elsewhere (Gray et al., 2014), although their numbers are increasing rapidly in line with global targets agreed under the CBD. In a meta-analysis that included some 20 marine protected areas across the Americas (mainly in the Caribbean, Central America and at Northwestern South America), Edgar et al. (2014) concluded that effectiveness is related to good design, isolation by deep water or sand, durable management and compliance related to "no-take" (or no fishing in specific zones or specific moments in the year). Other studies in the region relate effectiveness to environmental zoning, management plans, and participatory management (e.g. state of Ceará, Brazil: Andrade & Soares, 2017) or to network management (state of California, USA: Mach et al., 2017) (see also 6.3.2). Another peculiarity of marine protected areas is related to size. Most marine protected areas are relatively small in size (global median of 3.3 km^2) and the current expansion on the creation of large marine protected areas, led Ban et al. (2017) to investigate the social and ecological effectiveness of marine protected areas. After examining 12 large marine protected areas, three of them in the Americas (Galápagos, Ecuador: 133,000 km^2; Seaflower, Colombia: 65,000 km^2; and Central California National Marine Sanctuaries: 27,645 km^2), they found that effectiveness was related to age of the marine protected area, enforcement and, again, participatory processes. Nevertheless, considering both small and large marine protected areas, Davidson and Dulvy (2017) demonstrated that shortfall in marine protected areas remains significant when it comes to the conservation of endangered species. By using systematic conservation planning to prioritize conservation actions for sharks, rays and chimaeras (class with the highest proportion of threatened marine species), they found 12 nations with more than 50% of imperilled endemics, four of which are in the Americas: Colombia, Brazil, Uruguay, and Argentina. Among those, they found that Brazil and Argentina have low conservation likelihood (an index built based on 10 national measures including governance, economics and welfare, fishing, and human pressure).

Indigenous peoples' and community conserved territories and areas are found around the globe and have a long-standing history in the Americas. Under this umbrella, many types of areas exist: indigenous territories, community forests, sacred natural sites, community-managed coastal and marine areas, among others. They may cover at least as much area as non-ICCA protected areas do, help sustain ecosystems and services, and are the basis of livelihoods for millions of people. They are seen as efficient instruments to mitigate (Ricketts et al., 2010) and adapt (Magrin et al., 2014) to climate change and to reconcile biodiversity conservation with human development (e.g. Argentinian Chaco: Marinaro et al., 2015; Bolivian Andes: Hoffmann et al., 2011; Panama: Oestreicher et al., 2009). There is an inconclusive discussion as to whether protected areas without people inside or protected areas with people inside are more effective at promoting conservation. For instance, a comparative meta-analysis for reserves in different parts of the world, most of which in the Americas, showed that protected areas without people inside have higher deforestation rates than areas under community management (Porter-Bolland et al., 2012). Similarly, Nelson and Chomitz (2011) found for Latin America and the Caribbean that (1) protected areas of restricted use reduced fire substantially, but multi-use protected areas are even more effective; and (2) in indigenous reserves the incidence of forest fire was reduced by 16% as compared to non-protected areas. On the other hand, Miteva et al. (2012) found opposite results and suggested that fully protected areas are more efficient in constraining deforestation.
Despite mixed reviews, there are new, successful experiences in the Americas (Nygren, 2005; Lima & Pozzobon, 2005; Silvano et al., 2014) that show potential for enhancing biodiversity and ecosystem services conservation in the region. Some useful examples of adaptive community management include community forest concessions (e.g. Guatemala: Radachowsky et al., 2012), multiple-use management of forests (Guariguata et al., 2012; see also examples in Bolivia: Cronkleton et al., 2012 and Brazil: Klimas et al., 2012; Soriano et al., 2012); and local communities where payments are made to promote citizen collection of primary scientific data (Luzar et al., 2011). One of the main critiques of community-based management has to do with scalability, since many such local successes do not operate well at larger levels (Berkes, 2006). Moreover, one must consider that the ecosystems currently called 'native' have probably been, to some

extent, managed by humans, as the work by Levis et al. (2017) on the effects of pre-Columbian plant domestication over the structure of tree communities in Amazonia indicates. Another important related aspect is the relevance of indigenous peoples and local communities to conserving agrobiodiversity. These populations provide a largely under-recognised contribution to *in-situ* conservation and enhancement of crop diversity, as well as to high forest biodiversity, providing a free service that economists call positive externality (Carneiro da Cunha & Morim de Lima, 2017; Emparaire, 2017). Consequently, assuring the rights of indigenous and local populations to land and to keeping traditional management practices - inside or outside protected areas - is not only a matter of social justice; it is intimately related to a conservation strategy for biodiversity and ecosystem services conservation at a relatively low cost. This seems especially relevant for food security in the current global context of climate change, increasing population and an eroding genetic diversity of plant cultivars.

Biodiversity, ecological and conservation corridors provide connectivity and are essential to ensure flow of genetic material and ecosystem services (Hilty et al., 2006), despite the fact that they may not be equally efficient for different groups of species (Snäll et al., 2016). Therefore, policy design to address such concerns takes place at landscape scale and applies corridors as a policy instrument that will frequently have complementarity to existing protected area and/or ICCA networks, and may need to include ecosystem restoration (see 6.4.1.2) as an implementation tool. It is an instrument that potentially links units of conservation to promote an integrated conservation system within productive landscapes, and it has recently been argued that they may also serve as carbon corridors under REDD+ schemes, based on studies conducted in the Amazon, specifically in the Guiana Shield (Jantz et al., 2014). IUCN (2007) reported Latin America as leading international connectivity efforts, since, up to that time, more than 100 corridors had been created in 16 countries. Moreover, more than 20 of these corridors were multicountry. Although only Bolivia, Brazil and Venezuela had, by then, specific national legislation enabling corridors, there are examples at sub-national level (e.g. Argentina, Ecuador).

Another multicountry example is the Mesoamerican Biological Corridor, launched in 1994 (IUCN, 2007). It covers 27% of Mesoamerican territory and encompasses 26 indigenous groups, all the major Mayan archaeological sites, and 368 protected areas. Finally, in North America, the most important initiatives are driven by NGOs that aim to achieve their goals through broad-based stakeholder processes. This includes collaboration with government authorities to secure support through conservation policy and public land management. The corridor initiatives centre on biodiversity conservation and wilderness concepts. The best-known North American continental scale initiative is the Yellowstone to Yukon Conservation Initiative, extending along 3,200 km of the northern Rocky Mountains from Wyoming to the Arctic Circle. It includes areas protected under the national legislation of Canada and the USA, as well as private lands (IUCN, 2007). As a result of these various initiatives, the connectivity between protected areas in the Americas – alongside with Africa – is high when compared to other regions, and the networks of countries such as Argentina, Brazil and Canada are important to promote continental connectivity (Santini et al., 2016).

Other effective area-based conservation measures are considered as conservation mechanisms by Aichi target 11. Other effective area-based conservation measures must contribute to both the quantitative and qualitative aspects of target 11, and have the potential to contribute greatly to elements such as representativeness and connectivity, and to contribute to conservation in relevant places such as Key Biodiversity Areas, especially in cases where protected areas are not an option. Key Biodiversity Areas are sites that contribute significantly to the global persistence of biodiversity, including Important Bird and Biodiversity Areas, Alliance for Zero Extinction sites, and similar networks (UNEP-WCMC and IUCN, 2016). Key Biodiversity Areas, Important Bird and Biodiversity Areas and Alliance for Zero Extinction sites derive from analyses of threatened biodiversity, restricted-range biodiversity, ecological integrity, biological processes, and irreplaceability, across genetic, species, and ecosystem levels (IUCN, 2016). For that, the Red List of species is an important tool. The Red List of species is a 50 year old tool put in place by IUCN, which is the most widely used global imperiled species list (Rodrigues et al., 2006; Schipper et al., 2008). The use and application of the list varies across the region, but mainly there are differences between national lists and the IUCN Redlist. In the USA (Harris et al., 2012), Brazil and Colombia (Brito et al., 2010), the IUCN Redlist has a longer list of endangered species than in the country's official lists.

With respect to the relationship between the redlist and the design of marine protected areas, Agardy et al. (2011) highlight the need for protecting the core habitat of threatened species, to avoid population decline as found in the 1990's for the vaquita (*Phocoena sinus*), a small porpoise endemic to the northern Gulf of California, Mexico. However, the IUCN mechanism has been criticized for falling short on capturing functional and phylogenetic diversity, for instance, in the case of Brazilian birds (Hidasi-Neto et al., 2013). Mace et al. (2008) provide a detailed analysis a description of IUCN methods and its potential limitations. Measures of phylogenetic diversity and evolutionary distinctiveness are becoming more widespread and can potentially help to circumvent existing limitations (Faith, 2016). Controversy also exists around marine fisheries in the relationship between CITES (Convention on International Trade in Endangered Species of Wild Fauna and Flora), FAO (Food and Agriculture Organization, which revises CITES criteria periodically) and nations on the relevance of listing on CITES species that are commercially exploited (Cochrane, 2015).

Following the same logic, IUCN is in the process of developing a RedList for threatened ecosystems. Current challenges include ecosystem classification, measuring ecosystem dynamics, degradation and collapse, and setting thresholds to define categories of threat. Examples of potential applications of Red Lists of Ecosystems in legislation, policy, environmental management and education in the Americas region are found in the province of Manitoba (Canada) – a law on threatened ecosystems - and in Venezuela, which has a National Ecosystem Redlist (Keith et al., 2015).

6.4.1.2 Ecosystem restoration

Ecosystem restoration is a practice that is becoming widespread across the region. It can serve different purposes, such as climate change mitigation and adaptation, or circumvention of biodiversity loss. It can also help promote the integrity of existing protected areas and ICCAs and create biodiversity corridors. Furthermore, a meta-analysis of studies on ecological restoration of agroecosystems (54 published papers, five in the Americas) has shown that this practice is generally effective and can enhance biodiversity and the supply of supporting and regulating ecosystem services in agricultural landscapes (Barral et al., 2015). Another meta-analysis (Crouzeilles et al., 2016) performed on 221 landscapes worldwide (>50% in the Americas) that have undergone forest restoration found that it enhances biodiversity by 15–84% and vegetation structure by 36–77%, compared with degraded ecosystems. These authors also found that the main ecological drivers of forest restoration success are the time elapsed since restoration began, disturbance type and landscape context. Therefore, success in restoration efforts can help to positively influence the status of conservation of species, habitats and ecosystems (see also Chapter 4). However, there is still a limited repertoire of studies and practical actions related to ecological restoration policy and practice (Aronson et al., 2010; Baker et al., 2013). For instance, Jørgensen et al. (2014) showed that only three out of 58 articles in restoration-related journals globally identified specific policies relevant to their research results. Nevertheless, the increasing relevance of ecosystem restoration in the international agenda - such as the CBD; Aichi target 15 (Jørgensen, 2015; Murcia et al., 2016), or the Bonn Challenge (Liu et al., 2017) – can impact national policies. For instance, the Nationally Determined Contributions of Brazil to the Paris agreement of the United Nations Framework Convention on Climate Change (UNFCCC) is echoed by recently designed national restoration policies (Scarano, 2017).

Globally, there is an apparent emphasis on restoration efforts related to forest, coastal and freshwater ecosystems as opposed to dry and semiarid ecosystems (Aronson et al., 2010; Crouzeilles et al., 2016), which is a pattern mirrored in the Americas. Aronson et al. (2010) also indicate that restoration efforts tend to be more common in high income than in low income countries. In the Americas, however, restoration is already present in national legislation and policies of several countries, such as the USA (Baker et al., 2013; Palmer and Ruhl, 2015), Brazil, Colombia, Ecuador (Murcia et al., 2016) and Mexico (Ceccon et al., 2015). There is also a wealth of emerging bottom-up restoration initiatives, such as new international and national associations and collaborations including both practicioners and academics, in Latin America and the Caribbean (Echeverría et al., 2015).

There is a debate on the links between restoration science and policy, including legislation. Palmer and Ruhl (2015) argue that the USA legal system fails to distinguish between ecosystem restoration and any other type of environmental intervention, which, they argue, may imply continuation of net ecological losses. However, there are relevant national policies such as the Estuary Restoration Act that created a federal interagency (the Estuary Habitat Restoration Council) that leverages resources and expertise from different agencies to help restoration practitioners, such as local and state agencies, tribes and nongovernmental organizations (Schrack et al., 2012). More recently, a policy support tool has been developed that integrates network analysis results with ecological habitat data to subsidize a socioecological restoration planning for USA estuaries (Sayles & Baggio, 2017). In Mexico, Ceccon et al. (2015) claim that despite the existence of legal instruments at a national level to regulate ecosystem restoration, there are no specific instruments defining basic concepts, criteria and standards, required actions, or regulations to implement and evaluate ecological restoration. In Brazil, there was subnational legislation in the state of São Paulo that imposed high species diversity for restoration, which for many academics was a misinterpretation of the best science available (Durigan et al., 2010; Aronson et al., 2011). This eventually led to the legislation being overruled and being replaced by a new legal instrument that assesses success of restoration projects based on: ground coverage with native vegetation, density of native plants spontaneously regenerating, and number of spontaneously regenerating native species (Chaves et al., 2015).

Given the inherent high costs of ecosystem restoration, it can be promoted by economic incentives such as PES (Bullock et al., 2011) and/or by biodiversity-offset policies (Maron et al., 2012). These topics are discussed next (6.4.2). Clearly, however, the costs of restoring are much higher than the costs of conserving (Chapter 4). Thus, in cases where old-growth mature forest remains, such as in the Amazon, restoration of degraded areas is less of a priority than avoiding further deforestation (Fearnside, 2003). Finally, another type of ecosystem intervention that might have regulatory backing in some cases is the control and eradication of invasive alien species, which is explored in Box 6.2.

6.4.2 Incentive mechanisms

6.4.2.1 Conservation incentives

The global leaders' commitments to SDG aim at a sustainable use of the oceans, seas and marine resources as well as the protection of life on land including the sustainable management of forests, and halting biodiversity loss (SDG 14 and 15) (UN, 2015b). Furthermore, there is a clear need stated in Aichi target 17, related to the development, adoption and early implementation by 2015 of a policy instrument aligned with each signatory's national biodiversity strategy and action plan (CBD/UNEP, 2010). Given the level of political uncertainty and governance changes, it is strategic to seek for additional policy instruments to fund biodiversity and ecosystem conservation and its range of benefits. This is even more important, considering the severe under-funding of protected areas and the high costs of restoration, yet evident high value provided by nature and its benefits to people.

Conservation incentives are increasingly implemented as complementary and allegedly cost-effective means to align biodiversity and ecosystem services conservation efforts with a good quality of life in the Americas and elsewhere (Magrin et al., 2014). Such incentives are an alternative to governmental command-and-control measures, are contingent upon defined environmental outcomes, and are intended to encourage the adoption of more sustainable land uses. Different from the "polluter pays" principle, which is based upon land users bearing compliance costs, conservation incentives enable mechanisms where the beneficiaries compensate the providers for the additional provision or maintenance of desired ecosystem services (e.g. regulation of freshwater quantity). Examples of conservation incentives in the Americas include Payment for Ecosystem Services, REDD+, environmental certification, conservation easements, as well as sustainable finance instruments.

Payment for ecosystem services schemes are one of the most common examples of conservation incentives in the Americas. The focus of a PES scheme varies according to their purpose. For example, the main focus of PES is often to support and improve ecosystem management (especially related to carbon sequestration and storage, watershed protection, landscape aesthetics, and biodiversity protection), although some PES schemes also attempt to achieve multiple goals (i.e., poverty reduction, regional development or political objectives) (Rodriguez-Osuna, 2015). PES worldwide and in the Americas primarily target water-related ecosystem services (Ezzine-de-Blas et al., 2016). Amongst all types of PES, payments for watershed services and reciprocal agreements for water, known also as water funds, are becoming the most significant incentive-based tool for watershed conservation in Latin America (Goldman-Benner et al., 2012; Martin-Ortega et al., 2013; Rodriguez-Osuna, 2015; Grima et al., 2016).

Grima et al. (2016) identified 40 cases of PES across Latin America and they concluded that successful PES programs have in common four features: (1) the way ecosystem services are traded (i.e., securing the continued provisioning and quality of a critical resource while positively contributing to local livelihoods); (2) spatial and time scales are more likely to succeed (local and regional schemes with a duration between 10-30 years); (3) transaction types: in-kind contributions are preferred as opposed to solely using cash payments; and (4) successful schemes tend to involve mostly private stakeholders, and no intermediaries between buyers and sellers. Other authors (Wünscher et al., 2008; Southgate et al., 2009) highlight that PES program efficiency increases (especially in the case of budget constraints) when payments reflect differences in opportunity costs, transaction and direct protection costs, all of which vary across space. Furthermore, a meta-analysis of 55 PES schemes worldwide found that there are three key factors contributing to the likelihood of a PES scheme to have environmental additionality: 1) spatial targeting of contracts focused on hotspots of high ecosystem service intensity or high threat; 2) differentiated payments that consider variable provision costs across providers; and 3) to a lower extent, the degree of conditionality which refers to the implementer´s ability to monitor and sanction non-compliance (Ezzine-de-Blas et al., 2016). However, Börner et al. (2016) on a global synthesis including various studies across the Americas showed that the effectiveness of forest conservation instruments in the same category, including incentive mechanisms, vary greatly and suggest that it is too early for generalizations about pre-requisites for success.

Factors that were common across PES schemes that had a low degree of success include: schemes that did not reduce pressure on ecosystems, investors were not convinced of the impact of their investments, opportunity costs were not met, local livelihoods were not improved, land tenure arrangements and power structures were weakened and showed an unfair distribution of benefits (Grima et al., 2016). Other additional factors that might negatively impact the feasibility and implementation of PES schemes are: a perceptions of commoditization of nature, schemes that are not able to achieve poverty reduction, slow or absence of trust building between service users and providers and gender and land tenure issues (Asquith et al., 2008; Balvanera et al., 2012; Magrin et al., 2014).

Box 6.2. Control and eradication of invasive alien species

At the international level, the eradication of invasive alien species is one of the priority targets for the CBD's Aichi targets. Invasive alien species are the second greatest agent of species endangerment and extinction after habitat destruction (Pejchar & Mooney, 2009; see also Chapters 3 and 4), and drive economic setbacks. Indeed, this is a true international issue, since historically and up to this day, many species invasions are brought about by an indirect effect of trade or economic flows between countries and regions (Liu et al., 2013). Therefore, regarding coordination across jurisdictional boundaries, one of the CBD guiding principles on invasive alien species is cooperation (https://www.cbd.int/decision/cop/?id=7197), which suggests efforts related to information sharing and agreements (including on trade) between countries.

However, solutions are on most occasions addressed at the local level, where the impacts are most felt. For instance, in the USA, where many national and sub-national policies and research regarding invasive alien species are in place (e.g. Crowl et al., 2008; García-de-Lomas & Vilá, 2015), there is evidence showing that even small amounts of cooperation to control bioinvasions between neighboring individuals or groups can provide large social benefits. Therefore, coordination among managers across jurisdictional boundaries can have profound effects on the outcomes of invasive alien species management (Epanchin-Niell & Wilen, 2014). This conclusion also applies to transboundary multinational relations. For instance, in the region, North America has several mechanisms of collaboration between countries (Canada, USA, and Mexico), such as the North American Plant Protection Organization (http://www.nappo.org/; under the International Plant Protection Convention): the Forest Insects & Disease and Invasive Plants Working Group of the North American Forestry Commission (under the umbrella of the FAO; http://www.fs.fed.us/global/nafc/insects/ aboutus.htm) and the North American Invasive Species Network (http://www.naisn.org/). Fonseca et al. (2013) describe a tri-national initiative created to face the challenge of the invasive alien species associated with the South American Pampas, a grassland vegetation type that covers parts of southern Brazil, Argentina and the whole of Uruguay. They suggest that regional legislation to manage invasive alien species should be designed. At present, however, in the case of Latin America, national invasive alien species policies (Speziale et al., 2012) and research (Gardener et al., 2012) are less well developed. Existing national policies mainly deal with alien species threatening productive systems (Speziale et al., 2012).

The harmful effects of invasive alien species can be particularly serious in islands (Simberloff, 2011), and are a concern in the Caribbean. Since the beginning of this decade, The Bahamas, Dominican Republic, Jamaica, Saint Lucia and Trinidad and Tobago either revised or started developing national invasive alien species strategies, as well as regional invasive alien species strategies for freshwater, marine and terrestrial ecosystems. Challenges to success include shortage of scientific data, trained personnel and public awareness, insufficient coordination and collaboration, ease of introduction and movement of invasive alien species and inadequate quarantine facilities, and inadequate funding (GEF/UNEP/CABI, 2011). In Cuba, invasive alien species is dealt with by policies such as the National Environmental Strategy, the National Biodiversity Strategy, the Environmental Regulation System, and the Biosafety Regulations System. Since 2012-2013, a specific National Strategy has been developed and launched, which produced risk assessments and the elaboration of a List of Harmful Alien Species. In other Caribbean countries, invasive alien species prevention control, management and eradication has often been related to National Biodiversity Strategies and Actions Plans (NBSAPs) within the realm of the CBD. Finally, the Caribbean has launched a regional initiative in 2012-2013: The Regional Strategy for the Control of Invasive Lionfish in the Wider Caribbean. It has a broad participation from countries (including USA and Mexico), specialists and regional institutions, under the umbrella of the International Coral Reef Initiative. Collaboration on expertise exchange, integration of monitoring and legal alignment between countries increase effectiveness of this collaboration (Gómez Lozano et al., 2013).

A review of 190 publications on invasions by introduced mammals in southern South America forwarded a set of recommendations that are applicable to all invasive alien species (Ballari et al., 2016): to recognise the presence and spread of these species in pristine or protected areas; to improve controls to prevent new introductions and escapes; to include social and cultural aspects of biological invasions in research and management plans; to establish long-term programmes to monitor distribution and dispersion; to achieve societal involvement in management programmes to ensure public acceptance; and to develop prioritisation tools.

In Latin America, PES schemes are mostly publicly funded (65%) while the rest are private commercial and private non-commercial initiatives. North America shows a higher frequency of publicly funded PES (70%) (Ezzine-de-Blas et al., 2016). For example, payments made to preserve upstate New York watersheds that supply New York City with its water, while seemingly costly at $1-1.5 billion, are substantially less than the $6-8 billion needed to construct an additional filtration plant plus another $300-500 million in annual operating costs (Hanson et al., 2011). Another example is the so-called "Swampbuster program" of the USA

(Highly Erodible Land Conservation and Wetland Conservation Compliance Provisions) that financially incentivizes farmers to conserve wetlands (Geyer & Lawler, 2016).

Examples of publicly funded national schemes that launched PES are Mexico and Costa Rica. Since 2003, Mexico implements a federal PES program remunerating communities for forest conservation (now called PRONAFOR) that represents a significant additionality (12-15% of forested area protected) to conservation (Costedoat et al., 2015; LeVelly et al., 2015). Since 1997, Costa Rica's National Fund for Forest Financing has provided incentives for farms that provide upstream watershed protection, and also for carbon sequestration, biodiversity conservation and landscape aesthetic features. The major source of funds for Costa Rica's PES is a national tax on fossil fuel utilization (Montagnini & Finney, 2011). Guatemala, with its Program of Forest Incentives, benefited 4,171 beneficiaries who planted 94,151 ha of forest and put 155,790 ha of natural forest under protection by 2009 (INE, 2011). Ecuador launched *Socio-Bosque* in 2008 and by 2010 the program already included more than half a million hectares of natural ecosystems protected with more than 60,000 beneficiaries (De Koning et al., 2011). In Trinidad and Tobago, the Green Fund (established by law in 2004 by a national environmental fund) provides incentives for local communities and other non-business entities to undertake projects and programs that focus on: reforestation, restoration, conservation and education (UNEP, 2012). Peru has a program called "Conditioned Direct Transfers" that incentivizes sustainable production in the Amazon region, currently benefitting 57 native communities. Since 2014, Peru has also had a national law called "Compensation Mechanism for Ecosystem Services", which promotes voluntary agreements for water, carbon and biodiversity conservation.

Regional initiatives include Watershared (launched in 2015), which involves more than 125 local governments across the Andes of Bolivia, Peru, Ecuador and Colombia and uses reciprocal water agreements to preserve forests and to get downstream water users support upstream forest owners. This initiative includes 200,000-signed agreements benefiting around 4,000 families that conserve 200,000 ha of forests (Fundación Natura, 2016).

The private sector and businesses offer incentives (through different instruments such as PES and sustainable investments) for conserving and maintaining nature's contributions to society (Naturevest & EKO, 2014). This sector has shown a growing awareness about their increasing exposure to risks (e.g. to water scarcity, extreme weather events) as well as opportunities (sustainable investment) that can contribute to biodiversity and ecosystem service enhancement (UNEP FI, 2010; 2015; TEEB, 2010; WEF, 2015; WBCSD, 2017). For example, business incentives for ecosystem services include carbon sequestration and storage (e.g. through forest restoration activities). This ecosystem service is commonly traded in two types of markets where transactions for greenhouse gas emission reductions are in place. On one side, the compliance markets are ruled by the UNFCCC framework and on the other side, voluntary markets operate with requirements that are more flexible and where voluntary buyers drive demand. These markets allow companies, governments, NGOs and individuals to counterbalance their emissions through the purchase of offsets, commonly by certification of emission reduction credits (Rodríguez-Osuna et al., in press). The major share of voluntary carbon transactions came from REDD+ projects, which increased to nearly 50% of the total market share in 2013. In this context, Latin America has been the most important sourcing region, tripling their 2012 activity (Goldstein & Gonzalez, 2014).

Most of voluntary carbon transactions have been developed for the private sector, as well as local or international non-profit organizations and public entities. A significant value arose from Germany´s REDD Early Movers financing program in 2013 (government to government deal with the State of Acre in Brazil), which is a national program to avoid emissions based on performance. In this agreement, the State of Acre agreed to supply 8 million tons of carbon dioxide equivalent to the German Development Bank in 2013 (Goldstein & Gonzalez, 2014; Hamrick, 2015). Cumulatively, mostly voluntary offset supply came from the USA (136 million tons of carbon dioxide equivalent worth $656 millions), Brazil and Turkey (Hamrick et al., 2015). The most important buyer motivations for forest carbon transactions are corporate social responsibility (40%), demonstrating climate leadership (22%), compliance (17%), demonstrating industry leadership (13%) and taking action on climate change (12%) (Goldstein & Gonzalez, 2014).

Consumer preferences, investor's motivations as well as international agreements and agendas such as the SDGs and the Paris Agreement are increasingly driving the need for environmental certification and sustainable finance (Lewis et al., 2016). For example, the decline in deforestation in the Brazilian Amazon was partly driven by supply chain interventions, as in the example of the soy moratorium (see also 6.3.1, 6.3.4). Market opportunities for the business sector rise with increasing consumer preferences and demands for certification schemes for "climate neutral" products and services, sustainable fisheries, deforestation free products, among others (WEF, 2017).

Businesses can significantly foster conservation efforts and have a key role to play in halting biodiversity and ecosystem service loss as well as addressing the challenges posed by climate change. Already, the effects of climate change and the transition to a low carbon economy have progressively become a major feature driving adoption of corporate sustainable development strategies (KPMG, 2015). Most large global companies and many small and medium sized enterprises issues annual corporate sustainability reports (KPMG, 2013). The growing interest in sustainable finance is evidenced by several alliances and initiatives such as the Natural Capital Finance Alliance, with 90 financial institutions that have committed to collaborate towards understanding the natural capital risks and opportunities in their products and services (Redford et al., 2015; NCFA, 2016).

Conservation easements help protect private land from development and overuse, in exchange for a payment, tax reduction, or permit. The institutional context for conservation easements involve strained financial capacity, decentralized governance, and a mix of regulatory, incentive and market mechanisms (Rissman et al., 2015). They are particularly common in the USA and while they can significantly prevent habitat loss in agricultural regions (Braza, 2017), technical assistance and monitoring are frequently mentioned bottlenecks (e.g. Stroman & Kreuter, 2014).
According to a survey of 1200 chief executive officers worldwide in 2010, more executives in Latin America were concerned about biodiversity loss as a threat to business growth prospects than in North America (12%) (PwC, 2010). However, by 2016, the USA alone accounted for $8.7 trillion worth of sustainable assets under management integrated into their investment decisions (GSIA, 2016). In 2015, a well-supported investor initiative requested global major publicly listed companies to disclose their water and climate-related corporate risks. This request was made on behalf of 617 investors with $63 trillion worth of assets interested in climate corporate disclosure as well as 822 investors with $95 trillion worth of assets interested in water-related corporate disclosure (CDP, 2015a,b). These efforts indicate the growing opportunities for sustainable finance to fill the gaps needed to fund biodiversity and ecosystem's conservation (Naturevest & EKO, 2014).

6.4.2.2 Offset and compensation

Considerable progress has been made in developing good practice for biodiversity offsets (e.g. BBOP, 2012; Gardner et al., 2013), which are part of efforts to achieve no net loss of biodiversity while implementing development projects (Gardner et al., 2013). In the region they are particularly common in the USA and Canada (Coralie et al., 2015). Overall, there is a lack of integration of biodiversity and ecosystem services in impact assessment of large projects (mining, dams, roads), which set the scene for offset and compensation schemes (Brownlie & Treweek, 2013; Geneletti, 2016; Rodríguez-Osuna et al., 2017). Many of the historical problems with the application of biodiversity offsets have been assigned to lack of enforcement, poor governance, patchy monitoring, badly defined liabilities and lack of formal methods for designing and sizing offset requirements (Quétier & Lavorel, 2011). Indeed, it is often very difficult to consider all ecological dimensions of biodiversity (structural, functional, time, etc.) when offset calculations are made and final balance may fail to offset loss (Curran et al., 2014). They involve complex issues, even from an ethical viewpoint, such as the notion of species expendability (e.g. Kareiva & Levin, 2003). Gonçalves et al. (2015) in an extensive literature review found conceptual (choice of metric, spatial delivery of offsets, equivalence, additionality, time-scales, longevity, ratios and reversibility) and practical challenges (compliance, monitoring, transparency and timing of credits release) to biodiversity offset that deserve scientific attention. Furthermore, they argue that biodiversity offset locations could contribute towards a global network of biodiversity monitoring sites. On the other hand, Coralie et al. (2015) have a more skeptical view and argue that biodiversity offset discourse is framed by an economic rethoric resulting from political influence rather than by scientific robustness. In one point all these authors agree: more research is urgently needed to strengthen the evidence base on ways to achieve no net loss (Coralie et al., 2015; Gardner et al., 2013; Gonçalves et al., 2015). The concerns of climate scientist with the need for a "science of loss" (Barnett et al., 2016) also applies to biodiversity and ecosystem services. While biodiversity offset research can increasingly address damage, it remains challenging to address human losses that might derive from biodiversity loss, such as natural landscapes, cultures (such as those of indigenous peoples), and social cohesion (such as belonging to a community of knowledge or practice related to biodiversity and ecosystem services).

There is a rather uneven distribution of biodiversity offset studies in the region, which suggests a similar unevenness in practical application of this instrument. In a survey of 477 papers published globally between 1984 and 2014, the USA produced 57%, Canada 6.5%, Colombia 2.3%, and Brazil and Costa Rica with less than 1% - these were the only representatives of the Americas (Coralie et al., 2015). Gelcich et al. (2017) found similar results. However, it has been argued that while most offset research occurs in the USA, the majority of offset policies and programs are occurring in many middle- and low-income countries (Villarroya et al., 2014). Elsewhere in the Americas, most countries have policies that enable biodiversity offset (Gelcich et al., 2017), and indeed in Latin America, Brazil, Colombia, Mexico and Peru even explicitly require their implementation (Villarroya et al., 2014). The near absence of published papers or case studies on biodiversity offsets by Latin America and the Caribbean authors is a clear gap, given the intensive productive sector, the rich biodiversity, and the existence of enabling policies in most countries. This unevenness is also perceived between ecosystems. Most studies are concentrated on wetlands (Gelcich et al., 2017), perhaps because of the recommendation in the early 1970´s of the Ramsar Convention to compensate for damage to biodiversity (Hrabanski, 2015). The USA is also predominant in such types of studies on wetlands (Gelcich et al., 2017). However, Matthews and Endress (2008) reviewed monitoring information for 76 wetlands constructed between 1992 and 2002 in the USA, and found several problems with the performance criteria used to measure progress and assess compliance. Thus, some argue that wetland offset programs in the USA have often failed to meet their objectives, and have a poor track record of effective implementation and monitoring (Gelcich et al., 2017). Coastal and marine ecosystems are the ones with fewer studies on offsets, both regionally and globally (Gelcich et al., 2017). The USA again the main exception with a large number of studies. For instance, Levrel et al. (2012) reviewed cases in coastal and marine ecosystems in Florida over a ten-year period and found problems related to methodology, monitoring and uncertainties related to time-scale. For another coastal case in the region, in Brazil there is a type of financial compensation for giving up activities such as fishing certain species (including shrimp and lobster) during reproductive season, both in coastal and continental waters, which is called *defeso* (Begossi et al., 2011). On the positive side, the program by 2011 had benefitted nearly 650 thousand people, but the negative side of it is that there is evidence that funds have been transferred to people who are not involved in fishing activities (Campos & Chaves, 2014).

Brazil has recently introduced a new mechanism, known as Environmental Reserve Quotas that are tradable pieces of native or regenerating native vegetation, which are additional to what is required by the Brazilian environmental legislation. This mechanism allows landowners to offset surplus and deficits of legal reserve (minimum area required by law to be forest in private properties) and thus provide incentives to comply with the Native Vegetation Protection Law (Soares-Filho et al., 2014, 2016; May et al., in press).

There have also been advances on valuation and assessment of the economic component of biodiversity and ecosystem services, with regional-, national- or ecosystem-level application of tools such as The Economics of Ecosystems and Biodiversity in Business and Enterprise (Bishop, 2012; Kumar et al., 2013) and the World Bank's Wealth Accounting and the Valuation of Ecosystem Services. These tools have challenges related to interpretation and direct application to policy (e.g. Ring et al., 2010; Spangenberg & Settele, 2010; Bartelmus, 2015). In the region, their influence on decision-making is still reduced in some countries (e.g. The Economics of Ecosystems and Biodiversity in Business and Enterprise in Brazil - Roma et al., 2013; the World Bank's Wealth Accounting and the Valuation of Ecosystem Services in the Caribbean - Waite et al., 2015), while in others such types of approaches have often been incorporated to policies (e.g. USA – Schaefer et al., 2015).

6.4.2.3 Eco-certification and other mechanisms related to markets and trade

The local, subnational, national, intraregional and international trade is often regulated by policies, including incentives, disincentives and subsidies applied by each country and subregion. They can have a large impact, positive or negative, on biodiversity and ecosystem services. State's capacities to govern for sustainability are challenged by processes of globalization, such as the telecoupling (Lenschow et al., 2015; Chapter 5). For instance, Lenzen et al. (2012) have shown that 30% of global species threats are due to international trade, and that consumers in developed countries cause threats to species through their demand of commodities that are ultimately produced in developing countries. Many developed countries cause a larger biodiversity footprint abroad than at home, due to the consumption of imported coffee, tea, sugar, textiles, meat, fish, timber, extractives, and other manufactured items. In the world ranking of net importers of biodiversity threat, the USA is first and Canada is ninth. Honduras is among the main net exporters.

Trade linkages between producers of commodities (e.g. soybean, coffee, palm oil, paper and pulp and beef) and distant consumers have turned the Americas a key world commodity producer (see Chapters 1 and 4), which involves a significant ecological footprint (Moran & Kanemoto, 2017). World's total food and grains exports have increased tenfold in the past couple of decades. More than 80% of soybeans used by China's food industry are imported from Brazil and the USA. The soybean trade between these countries plays a major role in global trade markets and prices, carbon emissions, ecosystem services, and livelihoods in many coupled human and natural systems in China, Brazil, and beyond (Liu et al., 2013). In this context, eco-certification has emerged - partly due to consumer preferences and public legislation in industrialized countries that demand standards to ensure food safety and environmental sustainability in food (Garrett et al., 2013; Lambin et al., 2014; WEF 2017), or timber (Polisar et al., 2017), or mining production (e.g. Ribeiro-Duthie et al., 2017), among others. In the Americas (see also Table 6.2) and elsewhere, eco-certification is managed by governments (e.g. United Sates Department of Agriculture's organic certification in the USA), research institutions (e.g. Smithsonian Center's Bird Friendly Coffee), NGOs (e.g. Rainforest Alliance), multiple stakeholders (e.g. Forest Stewardship Council) or individual companies (e.g. Starbucks; C.A.F.E Practices) (Lambin et al., 2014).

The Marine Stewardship Council is a non-profit organization that delivers the most widespread fisheries certification program. However, there are only 10 Marine Stewardship Council -certified fisheries in Latin America and the Caribbean, 4% of the total number of certified fisheries globally (Pérez-Ramirez et al., 2015). These authors argue that Latin America and the Caribbean certified fisheries have good performance indicators for stock status, governance and management systems, and that shortage of information and high costs prevent more fisheries to adhering to the certification scheme in the region. Globally, however, the scheme has received some criticism regarding sustainability of the target fish stock, low impacts on the ecosystem, and effective responsive management (Christian et al., 2013).

Despite criticism in some fronts, several authors argue for the high potential additionality and low risk of leakage associated to eco-certification schemes (Lambin et al., 2014). Such schemes can create economic incentives linked to monitoring and enforcement efforts to deal with externalities caused by commodity production such as deforestation, soil erosion and agrochemical pollution. 'Bat-Friendly Tequila' brands were introduced in 2014 - as a result from a Mexico-USA partnership comprising scientists, tequila producers, responsible bartenders associations – and contributed significantly to bat conservation efforts. The lesser-long bat was declared 'endangered' in 1994 in Mexico and in 1998 in the USA but now its population has recovered, resulting on its removal from the endangered lists in 2015. This certification promotes blue agave (raw material for tequila) crops to blossom naturally, allowing these plants to be pollinated by bats, which in return make crops more diverse and healthy. A growing environmental awareness and public demand motivates growers, which in 2016 produced 300,000 bottles in five brands reaching Mexican and USA markets (Trejo-Salazar et al. 2016).

The effectiveness of eco-certification varies widely and depends on the ability to enforce standards, exclude unsustainable producers, generate price premiums or other economic rewards that are sufficiently high to certify farmers (Lambin et al., 2014). The few studies that measured farmer-level benefits from eco-certification found reduced economic benefits but key social and environmental impacts under favourable conditions, especially in the certified coffee production (Mas & Dietsch, 2004; Blackman & Rivera, 2011; Blackman & Naranjo, 2012; Rueda & Lambin, 2013). Overall, there is a need for improved evaluation of the effectiveness of eco-certification (Lambin et al., 2014; Tayleur et al., 2017). In addition, to promote lasting impacts, eco-certification shall strengthen institutions and partnerships on the demand side and ensure that farmers are compensated for the added costs associated with certification on the supply side (VanWey & Richards, 2014).

Environmental bonds are still another type of market mechanism. It is a deposit-refund system that secures environmental restitution in cases of high impact and harmful practices (Gerard, 2000; Boyd, 2002). This instrument can incentivize land users, industries, and companies to improve monitoring and management systems, and could be based on the potential loss of the environmental services and relative risk of possible damages (Garcia et al., 2017).

Other examples of subregional policies and entities regarding the regulation and freedom of markets and perceived results are summarized in Box 6.3 (section 6.5).

Table 6.2. Certification schemes for soybeans, coffee and cattle in the Americas including the type of eco-certification and a brief summary of what is certified, by whom it is managed and the countries where the certification is found.
Countries are (1) Argentina, (2) , (3) Bolivia, (4) Brazil, (5) Canada, (6) Chile, (7) Colombia, (8) Costa Rica, (9) Ecuador, (10) El Salvador, (11) Guatemala, (12) Mexico, (13) Paraguay, (14) Peru, (15) Puerto Rico, (16) USA.

Commodity	Certification type	Brief description	Managed by	Countries	References/weblinks
Soybean*	USA Soy Sustainability Assurance Protocol (SSAP)	Certification of Sustainability U.S. Soy based on participation in U.S. farm program	The U.S. Soybean Export Council (USSEC)	16	https://certification.ussec.org/
	Cert ID Non-GMO Soy Certification/ Proterra Standard	Cert ID certifies soybean that is not genetically modified and the Proterra standard is designed to demonstrate social responsibility and environmental sustainability based on the Basel Criteria on Responsible Soy.	Cert ID Europe Limited	4	https://www.cert-id.eu/Certification-Programmes/Non-GMO-Certification/Non-GMO-Soy-Certification
	Round Table on Responsible Soy (RTRS) Certified Soy	Certifies soy, derivatives and soy products along the supply chain, including flows of material and associated claims. RTRS is a global platform of stakeholders of the soy value chain, which aims to promote the production of responsible soy through cooperation and open dialogue with the parties involved for making it economically feasible, socially beneficial and environmentally appropriate.	RTRS Association	4	http://www.responsiblesoy.org
Coffee	Bird Friendly Coffee	Identifies and verifies that the produced organic coffee has been grown using shade management practices that provide good bird habitats.	Smithsonian Migratory Bird Center at the National Zoological Park	3,5,6,7,9,10,11,12,13,14,16	https://nationalzoo.si.edu/migratory-birds/bird-friendly-coffee
	C.A.F.E. Practices	Evaluates, recognizes and rewards producers of high-quality sustainably grown coffee for Starbucks stores, by examining the economic, social and environmental aspects of coffee production against a defined set of criteria.	Starbucks Coffee Company	1,2,4,5,6,10, 11,12,14,15	https://www.starbucks.com/responsibility/sourcing/coffee
Various	Rainforest Alliance	Ensures that a product (e.g. coffee, tea,	The Rainforest	5,16	http://www.rainforest-

	Certified	chocolate, fruit, ready-to-drink beverages and juices, flowers, paper and tissue products, furniture and more) comes from a farm or forest operation that meets comprehensive standards that protect the environment and promote the rights and well-being of workers, their families and communities.	Alliance		alliance.org/find-certified

*Even if certification schemes are in place, certified soybeans have only 2% of the global market share (WEF, 2017)

6.4.3 Rights-based approaches

Human rights and human dignity are key principles of the 2030 Agenda for Sustainable Development (UN, 2015b). Rights-based approaches are, therefore, essencial to be applied in initiatives related to nature conservation, sustainable use and sustainable development. Therefore, in the Americas, many policies and governance schemes have rights-based approaches as background. For instance, the delimitation of ICCAs (see 6.4.4.1), decentralization of natural resource management (Hajjar et al., 2017; see also 6.3.5), participatory processes (see 6.3.1) are types of initiatives and actions that must adopt rights-based approaches, and many of those have already been explored in this Chapter. This section examines two specific actions that take into account these principles: Access and benefit sharing and rights of Mother Earth.

6.4.3.1 Access and benefit-sharing

Despite broad agreement at the Conference of the Parties of the CBD, designing and implementing an access and benefit-sharing regime at the national level remains challenging. Lack of human and institutional capacities to grant access, monitor and negotiate mutually agreed terms, legal gaps, unrealistic expectations of quantity of monetary benefits, and ill-informed laws draft without the benefit of a multi-stake policy planning process, among other reasons, hinder the effectiveness and efficiency of the access and benefit-sharing mechanism (Chishakwe & Young, 2003; Correa, 2005; Glowka, 2000; Lewis-Lettington et al., 2006; Ten Kate & Wells, 2001). In the Americas, 15 countries signed the protocol and eight ratified (see Table 6.3). Experience in these countries often shows similar challenges, including lack of experience on the subject, issues with management and personnel, the existence of legal loopholes, complex and lengthy procedures, etc. The cases of signatories (Argentina, Colombia, Costa Rica, Cuba, Dominican Republic and Panama) and non-signataries (USA and Canada) are discussed next.

In Argentina, national legislation on access and benefit-sharing is at an early stage of development, and it currently lacks regulation on the requisites or procedure to obtain prior informed consent or the potential benefits to be negotiated or prioritized (Silvestri, 2015). Access and benefit-sharing legislation at the provincial level is incipient and found in eight out of 23 provinces that have passed regulations on the topic, and the cases of Tierra del Fuego and Jujuy are interesting in that they demand an agreement on scientific collaboration between local and foreign research institutions if genetic resources are to be accessed (Silvestri, 2015). Further north, in the Andes, the Andean Community established common rules on access and benefit-sharing for Bolivia, Colombia, Ecuador and Peru. In the case of Colombia, which has signed but not yet ratified the Nagoya Protocol, the national regime is the one given by the Andean Decision. Most important challenges are related to the fact that access and benefit-sharing legislation is mostly spread in different resolutions and, therefore, lack specificity. Nearly 130 access and benefit-sharing agreements – pursuing mainly non-commercial research purposes – have been signed between year 2003 and 2016 (Ministerio Ambiente Colombia, 2016), and until 2010 no monetary benefits resulted from this regime (Nemogá et al., 2010). Other points that require improvements are related to participation of local and indigenous communities (Nemogá, 2005), to the absence of specific procedures for obtaining prior informed consent, and to complex bureaucracy (Vargas Roncancio & Nemogá Soto, 2010). However, the recently lifted administrative obstacles that paralyzed the non-commercial research on genetic resources seem a leap forward, which will need to be evaluated in the future.

Panama was one of the first countries to regulate the access to its genetic resources in 1998 and to ratify the Nagoya Protocol. Panama's legislation requires a certificate of origin or provenance, issued by the national authority that serves as the legal recognition of the source or origin of the genetic material following the Nagoya Protocol's recommendation. This certificate of origin is required for patent applications based on Panama's genetic resources, irrespective of the country where the petition is filed (Cabrera Medaglia, 2013). Challenges include clarifying distinction between accesses to genetic resources for commercial vs. scientific purposes, protocols for access to indigenous knowledge, and the absence of a fund to manage the economic benefits derived from access and benefit-sharing contracts (Lago Candeira & Silvestri, 2013).

In Costa Rica, The National Biodiversity Institute has signed more than 60 bioprospecting contracts –all of which have been duly authorized by the national competent authority, and more than 180 permits have been issued between year 2005 and 2010 (Cabrera Medaglia, 2013). Costa Rica counts on a coherent and strategic legal framework. By law, benefits to be negotiated between Costa Rica´s National Biodiversity Institute and the user of genetic resources include a front payment that could amount to up 10% of the total research budget and is directly allocated to the National Ministry of Enviroment and Energy for biodiversity conservation. One setback in this process is that the monetary benefits are small (Richerzhagen & Holm-Mueller, 2005). Another setback is that as of 2014 the National Biodiversity Institute began to financially decline and collections were transferred to government (Fonseca, 2015).

In the Caribbean, both the Dominican Republic and Cuba are discussing draft regulations on access and benefit-sharing. In any case it is mentionable Cuba has established in 2011 the Cuban Intellectual Property Office. Such an office verifies the legality of the access to Cuban and other countries´ genetic resources in the framework of intellectual property rights petitions.

Canada has not developed yet a comprehensive access and benefit-sharing regime; however, it is working towards that goal at different levels within government. Today some laws and regulations at the federal, provincial and territorial levels cover some elements of access and benefit-sharing. In the USA, even though the country has not ratified the CBD or the Nagoya Protocol, rules for access to genetic resources located within national parks are in place. The regime does not differentiate between biological or genetic resources but it includes all scientific and development activities that may be performed on any specimens of biodiversity. The equalization has led to greater flexibility which in turn has a positive impact on the benefits gained by the country. Sale and commercial use of biodiversity elements are prohibited because their property belongs to the federal government; however, subsequent developments and knowledge generated from any specimens of biodiversity may be privately owned and negotiated. The access and benefit-sharing regime is not focused on the access to genetic resources per se, but on the subsequent potential commercial and industrial uses. Therefore, the cornerstone of the USA's access and benefit-sharing system is an agreement of intellectual property rights (Vargas Roncancio & Nemogá Soto, 2010).

Bioprospecting does not lead to biodiversity threat as long as it is undertaken under national and international access and benefit-sharing principles and regulations (Singh & Singh, 2015). However, the impact of access and benefit-sharing agreements in reducing biodiversity loss is yet to been seen, since such agreements are still limited and experience on the topic is poor, as discussed here. At this point in time, there is a range of impressions: from optimism with bioprospecting and access and benefit-sharing potential (Skirycz et al., 2016) to doubts about the potential of this value for maintaining large areas of tropical forest in Amazonia (Fearnside et al., 1999); and from optimism about the potential to conserve biodiversity (Richerzhagen, 2011) to the overall concerns with potential to promote social justice and biodiversity conservation (Martin et al., 2013).

6.4.3.2 Rights of Mother Earth

The rights of Mother Earth emerge from a cosmovision that, unlike the predominant anthropocentric western vision, perceives mankind and nature as one indivisible being (Pacheco, 2014). Bolivia and Ecuador are two examples of countries in the region that have affirmed these rights in their national legislations. For instance, in Bolivia, key components of their legislation includes: a) Right to life and the diversity of life; b) Right to stabilize concentrations of greenhouse gases in the atmosphere at a level that would prevent dangerous anthropogenic interference with the climate system, and in sufficient time to allow the components of Mother Earth to adapt naturally to climate change; c) Non-commodification of the environmental functions of Mother Earth; d) Right to support the restoration and regeneration capabilities of all its components that enables the continuity of life cycles; and e) Right to clean air and live without contamination (Pacheco, 2014). The concept of *buen vivir* – living well with oneself, living well with the community and living well with nature - is an essential piece in this cosmovision and set of rights. The case of Tungurahua, Ecuador, where the local community decided on a watershed management model rooted in indigenous norms, following buen vivir principles, is an example of how this instrument is put in practice (Kauffman & Martin, 2014; see also 6.3.5).

The examples of these countries are beginning to reverberate in other countries. For instance, in Brazil, for the first time in history, a river (Rio Doce, in the State of Minas Gerais), represented by an NGO (Associação Pachamama), has entered a lawsuit (available at https://docs.wixstatic.com/ugd/da3e7c_8a0e636930d54e848e208a395d6e917c.pdf) in the state´s capital city. It asked for the recognition of its rights to life and demanding a plan for disaster risk reduction for the local population in the watershed. This took place in November 5th, 2017, exactly two years after the river was victim of the worst environmental disaster in Brazil´s history, with the collapse of a dam and the spill of 40–62 million m^3 of mining tailings in the river (Garcia et al., 2017). In Colombia, Chaves et al. (2018) examined the adoption of the concept of *buen vivir* by a network of sustainability initiatives in rural areas. Although the authors believe the network still has some way to go as regards full accomplishment of *buen vivir*, they argue that they play an important role in articulating and promoting novel territorial relations. Building “territories of peace”, as the authors call them, will be a great asset for post-conflict reconstruction in the country.

6.5 Regional adherence to global policies related to biodiversity and ecosystem services

This section covers the participation of the countries in the region in the global conventions (CBD, UNFCCC) and also in regard to the SDGs. Box 6.3 shows especially multicountry agreements within the region and related entities.

Box 6.3. Global, regional and subregional cooperation agreements and/or entities, often related to trade, infrastructure or governance, and that address directly or indirectly sustainable development and/or biodiversity and ecosystem services.

Amazonian Cooperation Treaty Organization: it aggregates all Amazonian countries and has four strategic actions in its portfolio: “conservation, protection and sustainable use of renewable natural resources”, “indigenous affairs”, “regional health management, infrastructure and transport”, “tourism” and “emerging topics”. The latter includes climate change, regional development and energy). The topic “Conservation, protection and sustainable use of renewable natural resources” has six subtopis: forest, water resources, management, monitoring and control of wild fauna and flora species endangered by trade, protected areas, sustainable use of biodiversity and promotion of biotrade and research, technology and innovation in Amazonian biodiversity (ACTO, 2014).

Antarctic Treaty: it is responsible for governance of the Antarctic. Challenges include assessing financial penalties for environmental damage and regulating bioprospecting, the establishment of marine protected areas, international regulation of tourism (Kennicutt et al., 2014). Fifty-three countries are Parties to the Treaty, twelve of which are in the Americas: seven Consultative Parties (Argentina, Brazil, Chile, Ecuador, Peru, USA, Uruguay) and five Non-Consultative Parties (Canada, Colombia, Cuba, Guatemala, Venezuela). Arctic governance entities: there is a multitude of Arctic governance structures and Arctic scientific bodies (Depledge & Dodds, 2017), but efforts are underway to strengthen the Arctic Council - currently a nonregulatory forum - and the possibility of a United Nations Regional Seas Programme is being considered as a management tool for the Arctic Ocean (Fleming & Pyenson, 2017).

CAFTA-TR: agreement between the USA, the Dominican Republic, Costa Rica, El Salvador, Guatemala, Honduras, and Nicaragua that creates economic opportunities by eliminating tariffs, opening markets, reducing barriers to services, and promoting transparency.

CARICOM (the Caribbean Community): members (Antigua and Barbuda, The Bahamas, Barbados, Belize, Dominica, Grenade, Guyana, Haiti, Jamaica, Saint Lucia, St. Kitts and Nevis, St. Vicente and the Grenadines, Suriname and Trinidad and Tobago) aim to improve 1) standards of living and work, 2) employment of labor, 3) sustained economic development and convergence, 4) breadth of trade and economic relations with other States, 5) levels of international competitiveness, 6) organisation for increased production and productivity, 7) economic leverage and effectiveness of member States in dealing with other States and entities, 8) co-ordination of Member States' foreign and economic policies, and 9) functional co-operation (www.caricom.org). It promoted, in partnership with the Caribbean Community Climate Change Centre, major regional projects to strengthen institutional, national, and human capacities. CARICOM heads of Government adopted the Regional Framework for Achieving Development Resilient to Climate Change and mandated the Caribbean Community Climate Change Centre to develop a regional plan to implement this strategy, in addition to an investment program, a governance regime and a monitoring and evaluation system (ECLAC, 2013).

Fair trade: it approaches production and service chains looking for equitable sharing of income, improving conditions for consumers on every day shopping decision-making to cope with producers and service providers' better livelihoods. It is spread all over Americas and works with producers of banana, cocoa, coffee, cotton, flowers, sugars, tea, composite products, fresh fruit, gold, honey, juices, rice, spice and herbs, sports balls and wine (http://www.fairtradeamerica.org/Fairtrade-Products).

Green commodities: United Nations Development Programme´s Green Commodities Programme aims to improve the social economic, and environmental performance of agricultural commodity sectors of nations. The Programme works to improve rural livelihoods, mitigate climate change, and maintain ecosystem services and resilience of landscapes and seascapes. By targeting agricultural commodities that have high economic and political national relevance and are part of aggregated supply chains, the Green Commodities Programme optimizes the potential of public-private partnerships to support long-term sustainable change. The Programme aims by 2020 to contribute to enabling eight million farmers, managing 20 million hectares, to improve the sustainability of their practices and their livelihoods. There ongoing are public-private partnerships in Costa Rica, Dominican Republic, Ecuador, Honduras, Paraguay and Peru (UNDP, 2015).

Initiative for the Integration of Regional Infrastructure of South America (IIRSA)**:** South American countries agreement on joint action to further promote regional integration towards infrastructure investments. Members are Argentina, Bolivia, Brazil, Chile, Colombia, Ecuador, Guyana, Paraguay, Peru, Suriname, Uruguay and Venezuela (http://www.iirsa.org/Page/Detail?menuItemId=29). While some authors point out for the innovative nature of infrastructure financing of this international initiative (Vitte, 2011; Souza, 2015), others highlight the need for new environmental and social standards (Costa et al., 2015) or the risks of deforestation (De Lisio, 2013; 2014).

The Common Market of South (Mercosur), is a regional integration process initiated by Argentina, Brazil, Paraguay and Uruguay and which incorporated Venezuela and Bolivia. It is dedicated to create an open space for generating commercial opportunities and investments through competitive integration of national economies (http://www.mercosur.int/innovaportal/v/3862/2/innova.front/en-pocas-palabras).

The North American Free Trade Agreement (NAFTA), is the largest free trade region in the world, generating economic growth and helping to raise the standard of living for the people of Canada, USA and Mexico. NAFTA has benefited North American businesses through increased export opportunities resulting from lower tariffs, predictable rules, and reductions in technical barriers to trade. Along with increasing exports and imports, firms have become more specialized and thus more competitive (Canada, 2014; Villareal & Fergusson, 2017). The Commission for Environmental Cooperation was established in concert with NAFTA, the trade agreement, to foster conservation and to monitor and report on the impact of trade on the North American environment through the North America Agreement on Environmental Cooperation, NAAEC.

The Union of South American Nations (UNASUR): mechanism for the convergence of political and strategic objectives of Southern American countries and regional forum for conciliation, including over the environment (Silva & Brancher, 2014).

World Trade Organization (WTO)**:** agreements cover goods, services and intellectual property. They 1) spell out the principles of liberalization, and the permitted exceptions; 2) include individual countries' commitments to lower customs tariffs and other trade barriers, and to open and keep open services markets; 3) set procedures for settling disputes; 4) prescribe special treatment for developing countries; 5) require governments to make their trade policies transparent by notifying the Organization about laws in force and measures adopted, and through regular reports by the secretariat on countries' trade policies.

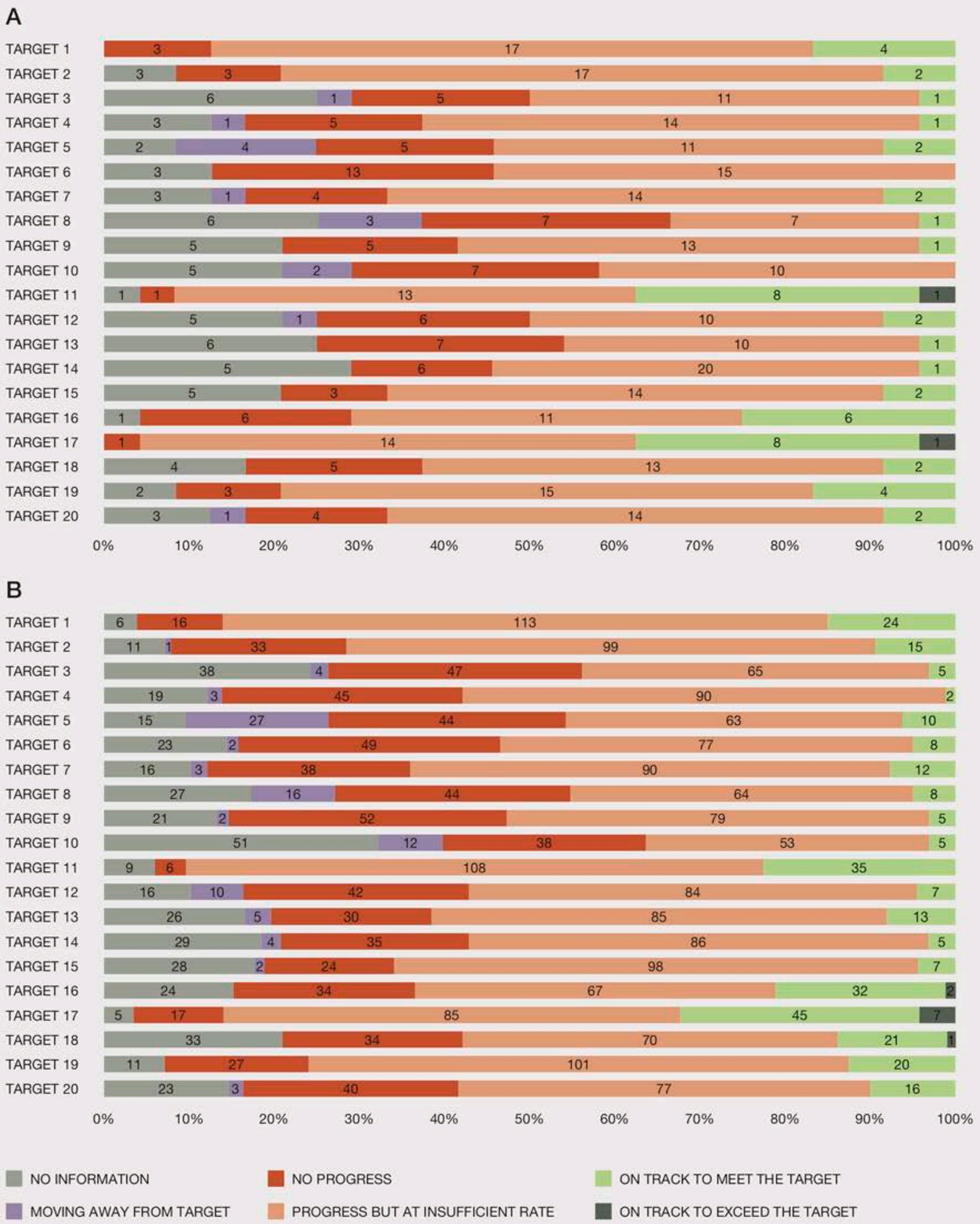
6.5.1 Convention of Biological Diversity

In the Americas, all countries, with the exception of the USA, have ratified the CBD agreement. So, next we will examine Latin America and the Caribbean countries and Canada.

Based on a mid-term assessment (2015) of the Strategic Plan, and with inputs from 24 countries in Latin America and the Caribbean, national governments of the Latin America and Caribbean region have thus far reported mixed results in progress towards the Biodiversity 2020 Aichi targets (UNEP-WCMC, 2016). Most progress has been reported in targets 11 (Protected areas) and target 17 (Adoption and implementation of policy instruments). There is evidence of good progress in target 1 (People aware of the value of biodiversity and the steps to conserve and sustainable use it); target 16 (Nagoya Protocol – section 6.3.2.5) and target 19 (Improved biodiversity information sharing). The targets most lagging behind however are targets 6 (Anthropogenic pressures/ direct drivers of change minimized) and 10 (Management of fish and aquatic invertebrate stocks).

Results from the 24 countries in Latin America and the Caribbean are summarized in Figure 6.3. Canada has similarly recorded progress in target 11 through the designation of several Protected Areas (Canada, 2014). Moreover, in Canada's biodiversity goals and targets (Canada, 2016) there is recognition of the relevance of traditional knowledge, innovations and practices of Indigenous communities for implementing all targets.

Figure 6 3 A summary of the status of accomplishment of Aichi Targets for 24 countries in Latin America and the Caribbean. Source: UNEP-WCMC (2016).



The Cartagena Protocol on Biosafety to the CBD, regulates the safe handling, transport and use of living modified organisms resulting from modern biotechnology that may have adverse effects on biological diversity, taking also into account risks to human health. Countries of America are 30 Parties (Table 6.3). All Latin America and the Caribbean countries are Party to the Cartagena Protocol on Biosafety, and some have a domestic regulatory framework fully in place (e.g. Brazil, Colombia, Saint Kitts and Nevis), others are partially in place (e.g. Bolivia, Costa Rica, Cuba, Dominique, Ecuador, Guatemala, Honduras, Mexico, Peru and Uruguay), while other countries do not have measures yet (e.g. and Dominican Republic).

6.5.2 United Nations Framework Convention on Climate Change nationally determined contributions

The USA, Canada, 17 Latin American countries (Argentina, Belize, Bolivia, Brazil, Chile, Colombia, Costa Rica, Dominican Republic, Ecuador, El Salvador, Guatemala, Honduras, México, Paraguay, Peru, Uruguay and Venezuela) and 15 Caribbean countries (Antigua and Barbuda, The Bahamas, Barbados, Belize, Cuba, Dominica, Grenada, Guyana, Haiti, Jamaica, Saint Kitts and Nevis, Saint Vincent and the Grenadines, Saint Lucia, Suriname, and Trinidad and Tobago) presented intended nationally determined contributions in the 21st Conference of Parties of the UNFCCC (Witkowski & Medina, 2016; Witkowski et al., 2016). Mostly, Latin American countries presented mitigation goals based on renewable energy and some with forestry. On adaptation the focus in on ecosystems, their conservation and services vulnerability risks, together with water management and efficient use. Caribbean countries have mitigation plans for energy, transportation, forestry, agriculture, industry, waste management and land use. Adaptation and building resilience are priority issues, the sectors identified as most vulnerable to climate change include agriculture, water, fisheries, tourism, human health, and coastal resources, as well as human settlements.

6.5.3 Sustainable Development Goals

The United Nations Rio+20 summit that took place in Brazil in 2012, committed governments to create a set of SDGs that would be integrated into the follow-up to the Millennium Development Goals after their 2015 deadline. In September 2015, 17 SDG, along with their 169 targets, were adopted by governments around the world as a part of the Global 2030 Agenda for Sustainable Development[33]. The SDG are important in the Americas context to help advance the progress already made under the Millennium Development Goals (such as the increase in the number of Protected areas). The SDG also provide an opportunity to address some of the persistent environmental challenges in the region (e.g. the net loss of forest in Latin America – especially South America – despite the development of a number of forest laws and policies across the region)[34].

[33]http://www.un.org/sustainabledevelopment/development-agenda/
https://sustainabledevelopment.un.org/post2015/transformingourworld
[34] http://www.mdgmonitor.org/mdg-progress-report-latin-america-caribbean-2015/

Table 6.3. Status of countries regarding two protocols from the Convention of Biological diversity: the Nagoya Protocol on Access and Benefit Sharing and the Cartagena Protocol on Biosafety.
Status S= signed; R= ratified; √ = under implementation

Country	Subregion	Nagoya	Cartagena
Antigua and Barbuda	Caribbean	R	√
Argentina	South America	R	
The Bahamas	Caribbean		√
Barbados	Caribbean		√
Belize	Mesoamerica		√
Bolivia	South America	R	√
Brazil	South America	S	√
Canada	North America		
Chile	South America		
Colombia	South America	S	√
Costa Rica	Mesoamerica	S	√
Cuba	Caribbean	R	√
Dominica	Caribbean		√
Dominican Republic	Caribbean	R	√
Ecuador	South America	R	√
El Salvador	Mesoamerica	S	√
Grenada	Caribbean	S	√
Guatemala	Mesoamerica	R	√
Guyana	South America	R	√
Haiti	Caribbean		
Honduras	Mesoamerica	R	√
Jamaica	Caribbean		√
Mexico	Mesoamerica	R	√
Nicaragua	Mesoamerica		√
Panama	Mesoamerica	R	√
Paraguay	South America		√
Peru	South America	R	√
Saint Kitts and Nevis	Caribbean		√
Saint Lucia	Caribbean		√
Saint Vincent and the Grenadines	Caribbean		√
Suriname	South America		√
Trinidad and Tobago	Caribbean		√
USA	North America		
Uruguay	South America	R	√
Venezuela	South America		√

Sources: Parties to the Protocol and signature and ratification of the Supplementary Protocol at https://www.cbd.int/abs/nagoya-protocol/signatories/default.shtml)
The Access and Benefit Sharing Clearing-House. https://absch.cbd.int/search/countries (Accessed November 22, 2017)

Although the SDG are not legally binding, governments are expected to integrate all 17 goals into their national planning frameworks. Two of the SDG relate directly to the management of biodiversity and ecosystem services: Goal 14 (Life below water: Conserve and sustainably use the oceans, seas and marine resources for sustainable development) and Goal 15 (Life on land: Protect, restore and promote sustainable use of terrestrial ecosystems, sustainably manage forests, combat desertification and halt and reverse land degradation and halt biodiversity loss). The remaining 15 goals are all considered relevant from a biodiversity and ecosystem services mainstreaming standpoint. Some goals are actions that are necessary to support/enable biodiversity and ecosystem services management (e.g. Goal 13 on Climate action and Goal 12 on Responsible consumption and production); while other goals can be supported and partially achieved (to varying extents) by effective biodiversity and ecosystem services management (e.g. Goal 3 on Good health and well-being and Goal 1 – No poverty).

The SDG are already showing signs of stimulating countries to identifying critical policy entry points for attending to the wide range of drivers affecting biodiversity and ecosystem services. While it is still too early to assess SDG impact on biodiversity and ecosystem services policies and management frameworks in the Americas, one example alights from Trinidad and Tobago. The country development of a new policy instrument – the National Spatial Development Strategy (which replaces the country's 1984 National Land Use Plan) – which has placed the protection of ecosystems at the core of spatial planning (Figure 6.4).

Figure 6 4 **The Sustainable Development Goals (SDGs) through the lens of biodiversity and ecosystem services mainstreaming.**

Schematic linkages amongst the SDGs from a biodiversity and ecosystem services (BES) standpoint. Source: Government of the Republic of Trinidad and Tobago (2016).

Given that CBD Parties continue to work towards the 2020 Aichi targets (section 6.4.1), the linkages between the SDG and the Aichi targets have been carefully considered so that efforts can be effectively aligned (Table 6.4). In many cases, the links between SDG and Aichi targets are strong and clear, and this implies that where governments have put measures in place to achieve the Aichi targets, there will already be progress under corresponding SDG.

Fulfilling the goals and targets of the 2030 Agenda for Sustainable Development will require a considerable effort to mobilize development financing, with both public- and private-sector involvement. In terms of domestic resource mobilization, one of the key challenges for the America's governments is raising the tax burden and improving the tax structure. This means addressing the problems of tax evasion and avoidance, both domestically and on the external front (ECLAC, 2016).

Table 6.4. Links between the Sustainable Development Goals and the Aichi targets. Source: CBD (2016).

Sustainable Development Goal	Relevant Aichi Biodiversity Target
1. End poverty in all its forms everywhere	2, 6, 7, 14
2. End hunger, achieve food security and improved nutrition and promote sustainable agriculture	4, 6, 7, 13, 18
3. Ensure healthy lives and promote well-being for all at all ages	8, 13, 14, 16, 18
4. Ensure inclusive and equitable quality education and promote lifelong learning opportunities for all	1, 19
5. Achieve gender equality and empower all women and girls	14, 17, 18
6. Ensure the availability and sustainable management of water and sanitation for all	8, 11, 14, 15
7. Ensure access to affordable, reliable, sustainable and modern energy for all	5, 7, 14, 15, 19
8. Promote sustained, inclusive and sustainable economic growth, full and productive employment and decent work for all	2, 4, 6, 7, 14, 16
9. Build resilient infrastructure, promote inclusive and sustainable industrialization and foster innovation	2, 4, 8, 14, 15, 19
10. Reduce inequality within and among countries	8, 15, 18, 20
11. Make cities and human settlements inclusive, safe, resilient and sustainable	2, 4, 8, 11, 14, 15
12. Ensure sustainable consumption and production patterns	1, 4, 6, 7, 8, 19
13. Take urgent action to combat climate change and its impacts	2, 5, 10, 14, 15, 17
14. Conserve and sustainably use the oceans, seas and marine resources for sustainable development	2, 3, 4, 5, 6, 7, 8, 10, 11, 12, 14, 15, 17, 19
15. Protect, restore and promote sustainable use of terrestrial ecosystems, sustainably manage forests, combat desertification and halt and reverse land degradation and halt biodiversity loss	2, 4, 5, 7, 9, 11, 12, 14, 15, 16
16. Promote peaceful and inclusive societies for sustainable development, provide access to justice for all and build effective, accountable and inclusive institutions at all levels	17
17. Strengthen the means of implementation and revitalize the global partnership for sustainable development	2, 17, 19, 20

6.6 Case studies highlighting cross-cutting issues in policy and governance

Four topics have been selected for a closer examination in this section – (1) Ecotourism, (2) Genetically modified crops, (3) Ecosystem-based adaptation to climate change and disaster risk reduction and (4) Science-policy interface. These studies seek to highlight some of the cross-cutting issues that have arisen in the assessment of policy and governance, and the include the following underlying considerations a) they have social, economic and environmental relevance all across the region; b) they link various points discussed in this Chapter, from policy mixes and instruments to cross-scale issues to regional integration; c) they represent large opportunities and/or involve major risks; d) science is largely inconclusive or unresolved about them; e) they represent significant science-policy communication challenges; and f) they point to knowledge and policy gaps, limitations and needs.

6.6.1 Case 1: Ecotourism

Travel and tourism industry generates nearly 10% of the GDP and 10% of all employment in the world**.** In the Americas, travel and tourism GDP is three times larger than auto manufacturing (WTTC, 2016). Possibly one billion tourists travel around the world every year (Stronza, 2008). Tourism is therefore one of the world's largest economic sectors and ecotourism is its fastest growing component (Klak, 2007). For instance, visits to protected areas in the USA and Canada have been estimated at 3.3 billion people per year (Balmford et al., 2015). Such numbers can be both promising and worrisome for nature conservation and socioeconomic sustainability. Clearly, a lot of the optimism in the 1990's about the potential positive impacts of ecotourism to nature conservation and sustainable development, has been challenged by negative views including the notion that tourism of any kind poses threat to nature conservation, and that revenues created by ecotourism are too small to support conservation on a larger scale (Krüger, 2005).

Some of the evidence is therefore conflicting or contradictory. For instance, while comparing 251 ecotourism case studies around the globe, Krüger (2005) found that ecotourism was perceived as less sustainable in South America, as well as in islands (such as in many places in the Caribbean) and mountain habitats. Moreover, even renowned ecotourism destinations, such as the Galapagos Islands, are challenged by socioeconomic issues associated with land-use practices and tourism management strategies (Durham, 2008). This result contrasts to the findings of Gunter et al. (2017) showing that ecotourism has a positive impact in poverty alleviation in 12 Central American and Caribbean countries. Indigenous or community-based ecotourism are indeed common across the region (e.g. Amazon: Gordillo-Jordan et al., 2008; Rodríguez, 2008; Canada: Williams & Peters, 2008; Panamá: Pereiro, 2016), and while some see it as sustainable and beneficial to local communities (e.g. Whitford & Ruhanen, 2016), others argue that it has often failed to deliver conservation and social benefits due to issues such as shortage of human, social and financial capital, lack of mechanisms for sharing benefits, and land insecurity (Coria & Calfucura, 2012). Such contrasting viewpoints can be related to a) the fact that evaluation is difficult, and starts from how an ecotouristic activity is defined (Buckley, 2009); and b) to intrinsic biophysical or political properties of localities or communities where the activity takes place.

In the Americas, there is a marked contrast between and within subregions. In Canada and the USA the annual economic impact of visits to protected areas alone is minimally $300 billion (Balmford et al., 2015). In Latin America and the Caribbean, Costa Rica, Belize, Bolivia, Ecuador, Mexico and Peru are popular destinations, while Cuba has an increasing potential and Brazil is currently developing its ecotourism potential (Borges Hernández et al., 2008). A comparative study between tourism in protected areas between Canada and Brazil is illustrative of how policies influence the impact of ecotourism on conservation and human wellbeing in and around protected areas. While these countries have two of the largest protected area systems in the world, policies are different: the Canadian focus is on leisure and recreation activities, while in Brazil it is on low impact activities (Matheus & Raimundo, 2016). Thus, revenues are larger in the Canada system and promotes local economies. In parallel, these authors argue, conservation is more related in both countries to monitoring and enforcement than to the presence or absence of visitors.

Rural Mesoamerica and the Caribbean contain other promising ecotourism landscapes and sites. These regions are therefore an interesting starting point for innovations on the topic. Klak (2007), by examining ecotourism in these two regions and especially cases in Mexico (the Monarch Buttefly Reserve), Costa Rica and Dominica, argued that ecotourism has synegies with cultural, historical, and agro-tourism, and that such components should be treated as a sustainable tourism ensemble, so as to combine ecological integrity, economic viability, and social justice.

Finally, the sustainability of ecotourism depends not only on implementation and management at the local level but also on large-scale external socioecological changes. For instance, disasters caused by extreme climatic or geological events (as seen during the past ten years in Brazil, Chile, Costa Rica, Guatemala, Haiti, Peru, USA; see also 6.5.3), social unrest (as seen in Mexico and Venezuela), and epidemic diseases (such as recently seen in Brazil and elsewhere in the Americas for dengue fever and zika virus), can lead to significant decline in tourism revenues (Pegas & Buckley, 2012; Woosnam & Kim, 2014).

Thus, the success of ecotourism depends on policies that mix nature conservation goals with income generation and, in given areas, poverty reduction. Participatory governance, social stability, and sociocultural ties with nature are also essential, particularly when community- or indigenous-based ecotourism is the case in question.

6.6.2 Case 2: Genetically modified crops

The use of genetically modified (GM) crops for many countries is an additional alternative within a range from more traditional to more industrialized production systems that include options (such as organic agriculture) that need to be maintained as viable options for food production (Burgeff et al., 2014). Doubts remain as to whether there are trade-offs between GM crops and organic farming (Azadi & Ho, 2010). Meanwhile, research, release in the environment, commercialization and the consumption of GM crops and associated products has been motivating debates regarding their potential benefits and possible risks. Although debates seem far from reaching consensus, many acknowledge: a) the potential of GM technology to increase food supply (at least in the short term), and b) the potential risks related mainly to fear of food safety and consequently, health and environmental impacts (Azadi & Ho, 2010; Hilbeck et al., 2015; Fahlgren et al., 2016).

This debate has been particularly intense in the Americas that by 2004 had 94% of the world's GM area (Traxler, 2006). To this day, the four world leaders in area of GM crops are the USA, Argentina, Brazil and Canada (Jacobsen et al., 2013). Out of these four countries, only Brazil has adhered to the CBD's Cartagena Biosafety Protocol (see Table 6.3). In Latin America and the Caribbean, only nine countries (Argentina, Brazil, Chile, Colombia, Cuba, Honduras, Mexico, Paraguay, Venezuela) have experience with biosafety regulatory activities, which indicates the need for regional partnerships in exchanging capacity and know-how (Rosado & Craig, 2017).

The controversy around GM use and the regulatory gaps across the region suggest that this topic should be discussed on a case-by-case basis at this point in time (Juma & Gordon, 2014).
The debate in the Americas is also intense because of the fact that the region has some of the main centres of origin of crop diversity in the world, in places such as Central America and Mexico; parts of the Andes, Chile and Brazil–Paraguay (Khoury et al., 2016). These countries and subregions have responsibilities regarding conservation and maintenance of the genetic pools of the crops represented in the wild relatives and landraces present within the country's boundaries, since any factor that might affect their integrity threatens genetic diversity for future global needs (Burgeff et al., 2014). This is reason for concern particularly in face of evidence of genetic erosion of such crop varieties in the aforementioned countries (e.g. Van Heerewaarden et al., 2009; Dyer et al., 2014). Burgeff et al. (2014) refer to the challenges related to coexistence of GM, conventional and organic options, especially for maize and cotton, in Mexico, which is a megadiverse country and also a centre of origin and genetic diversity for these two crops. They discuss the efficiency of the risk avoidance strategies such as setting distances between GM and non-GM producing fields to avoid pollen flow and genetic contamination, particularly if commercial releases take place. They also argue, although the guidelines for monitoring, verification and compliance of biosafety measures of GM crop releases have been established, that full implementation has not been achieved yet. Furthermore, this paper also examines the relationship between GM soybean and honey production in the Yucatan Peninsula: they claim that coexistence of these two activities in the same territory creates the risk of GM pollen as part of the honey produced. If so, Burgeff et al. (2014) argue that public perception may impact the acceptability by honey consumers, affecting the economy of thousands of rural people.

6.6.3 Case 3: Ecosystem-based adaptation to climate change and to disaster risk reduction

Adaptation requires capacity to allocate and to combine different types of resources for an uncertain future in a given place (Lemos et al., 2016). Thus, the deployment of ecosystem-based adaptive strategies and instruments shall vary depending on the local setting. Whether the locality in question is urban, rural, coastal, or in the middle of low populated wilderness areas - such as in most of the Amazon, for instance -

the adaptive strategy will vary largely (Scarano, 2017). This section will examine two adaptive practices and policies that are becoming common across the region but that still have major knowledge and policy gaps: ecosystem-based adaptation to climate change (EbA) and ecosystem-based disaster risk reduction (Eco-DRR).

Starting from the premise that places more vulnerable to climate change and natural disasters are those that lost their life supporting systems and that the people more vulnerable to such hazards are the poor people (Fisher et al., 2014; IPCC, 2014; Magrin et al., 2014), policies and practices that reduce poverty while protecting or restoring nature are adaptive. In the case of EbA, such actions should also mitigate carbon emission (Locatelli et al., 2011; Scarano, 2017). Although EbA and Eco-DRR have much in common, they bear some relevant differences that hamper communication and exchange between these two fields: they operate under different policy fora (climate change adaptation vs. disaster risk reduction), they address different types of hazard (climate vs. multiple), and they function under different time-spans (long-term vs. response, recovery and reconstruction) (Doswald & Estrella, 2015).

Ecosystem-based adaptation to climate change and Eco-DRR policies currently vary from global agreements to national adaptation plans to municipal strategies to local governance arrangements at smaller territories. However, global agreements do not necessarily percolate to national and sub-national policies, whereas local ecosystem-based approaches and solutions do not always scale up beyond the community that developed them (Scarano, 2017). In the international arena, barriers for mainstreaming it into climate policy are related to governance, effectiveness, time scale of processes, financing and scientific uncertainty (Ojea, 2015).

At national level, a review by Pramova et al. (2012) showed that only 22% of the National Adaptation Programmes of Actions 44 least developed countries incorporated ecosystem components. From the Americas, only Haiti was part of this study and its National Adaptation Programme of Action, suggests reforestation together with technical solutions (dry walls, gabions and stone lines) in several projects targeting watershed restoration for reducing the negative impacts of extreme climate events.

At sub-national level, the use of ecosystems in helping people adapt to climate change is limited by the lack of information on where ecosystems have the highest potential to do so, which can be at least partially overcome by spatial prioritization efforts (Bourne et al., 2016; Kasecker et al., 2017). However, whenever such policies already exist, approaches vary. Moreover, ecosystem-based approaches to climate change are not often systematized or labelled as such across the planet (Munroe et al., 2011). Similarly to policy actions, scientific literature on EbA is divided between global and local focus, and at local level it is divided according to the setting. Munroe et al. (2011) reviewed 132 papers on EbA and found that nearly half (45%) were from developing countries. They also found a predominance of papers focused on urban or rural, wetlands, forests and coastal ecosystems. However, a recent review on urban EbA covered 110 papers in 112 cities and there was a strong bias towards Europe and North America (Brink et al., 2016). Coastal vegetation, including mangroves, both in continental and in small island countries, have been highlighted as important for EbA (Mercer et al., 2012; Duarte et al., 2013; Martin & Watson, 2016), but still lack a thorough and integrated review effort. In rural areas, for smallholder farmers - especially based on studies located in Mesoamerica (Vignola et al., 2015) - EbA practices may improve the ability of crops and livestock to maintain crop yields and/or buffer biophysical impacts of extreme weather events or increased temperatures under climate change. In another review, Renaud et al. (2016) indicate that three major applications of Eco-DRR are conservation and management of a) coastal ecosystems for coastline protection (e.g. Chile - Nehren et al., 2016; USA – David et al., 2016); b) riverine ecosystems for floods protection (e.g. Argentina – Zimmermann et al., 2016); and c) protection of forests for landslide risk reduction (e.g. Brazil – Lange et al., 2016).

Estrella et al. (2016), reviewing experiences around the world, suggest that having an enabling policy, legal and institutional environment is needed to encourage implementation of Eco-DRR and EbA initiatives. They recognize, however, that due to the multi-disciplinary and multi-sectoral nature of ecosystem-based approaches, it becomes challenging to working with existing sectoral policies, and sectoral legal and institutional frameworks that often do not favour integrated approaches. Triyanti and Chu (2017) propose that future studies on governance systems for Eco-DRR and EbA should build operationalization strategies based on existing governance theories and methodologies, while also aiming for integrated assessments that evaluate socio-political, institutional, and power dynamics across different scales and political arenas.

6.6.4 Case 4: Science-policy interface

There are different perspectives on how efficiently scientific findings are translated into policy action. Some authors (e.g. Cáceres et al., 2016) envisage two models: the "information deficit model" (or deficit model or science deficit model) and the "power dynamics model". The first proposes that poor translation of scientific findings into policy implementation results from the lack of understanding of science or access to good data by decision-makers (e.g. Posner et al., 2016). In the second, science is just one additional element among many that are taken into account in the process towards an inherently political decision (e.g. Azevedo-Santos et al., 2017). Other authors propose that for science to have impact on policy it must have three properties: credibility, legitimacy and relevance (Sarkki et al., 2014). Sarkki et al. (2015) added iterativity to these three properties, and listed 14 features of the science-policy interface. Irrespective of the model used, fact is that policy makers use insufficiently the research-based knowledge available and researchers typically produce insufficiently knowledge that is directly usable (Weichselgartner & Kasperson, 2010). Here we examine three different cases at the science-policy interface and discuss how they relate to these different models.

For the Caribbean, Jacobs et al. (2016) examined empirical evidence of 130 conservation organizations in 21 countries to conclude that bridging the science-policy equals to bridging the knowing-doing gap. They argue that barriers to overcome this gap include lack of information and data sharing, political constraints, competition, limited resources and technical capacity, and ineffective communications. They claim that boundary organizations, i.e., groups that facilitate the transfer of knowledge between science and action can use the social sciences and humanities and practitioner expertise to successfully become knowledge brokers. Some Caribbean conservation organizations report that their greatest needs are not for more information but for capacity building in science and technology. They recommend that the focus should be on connecting the information to the appropriate users, providing support services to existing governance structures instead of developing new management frameworks and communicating the information in a way that users can understand and apply. Conservation partnerships are rendered ineffective when research results are not communicated to managers and translated to actions. Thus, in cases of failure in the science-policy communication, the Caribbean case seems to fit the Information Deficit Model, explained above, or the conservation science conveyed to policy makers may lack credibility, legitimacy or relevance.

The case of the Brazilian Official list of threatened plant species starts in 2009, when the publication of the list did not satisfy scientists or environmentalist, since the total number of threatened species was much smaller than that indicated by academics, based on IUCN criteria. Scarano and Martinelli (2010) interpreted that there was a large disagreement between scientists and policy-makers on the degree of scientific certainty required to define a species as threatened. The imbroglio surrounding the publication of the list was contemporary to a period of economic growth and high infrastructural investment in Brazil. Since Brazilian legislation strictly limits or forbids human activities in areas where threatened species occur, the national political stand was then somewhat conservative as regards threatened species conservation and environmental issues as a whole. Thus, decision-makers claim for higher certainty was hardly surprising. The turning point was the fact that, during negotiations for publication of the list, a National Center for Conservation of the Flora was created inside the Botanical Gardens of Rio de Janeiro. Five years later, the Center – with a better structured information database and scientific network established – updated the list from 417 (in the 2009 official list) to 2,118 (in the 2013 Brazilian Red List), and the official list, published in 2014, considered as threatened all 2,118 plant species proposed by the Center (Scarano, 2014). So, in this case, a Power Dynamics Model operated in the 2009 list, and was superseded by better information and communication in 2014, which fits better the Information Deficit Model. It can also be seen as a case where the first list lacked credibility from the policy-makers' perspective, given the different perceptions of uncertainty and risk that the listing method generated to different stakeholder groups. The second list resolved the different risk tolerances of scientists and policy-makers and agreement was reached.

Finally, the case described in the same paper by Cáceres et al. (2016) fits the Power Dynamics model. The case relates to the process leading to the Córdoba Provincial Law for the Protection of Native Forests, in Argentina. Viewpoints between actors of the agribusiness sector and of some political parties contrasted with that of environmental groups and campesinos organizations. The authors argue that the result was the expression of a power dynamics that disregarded scientific evidence. The authors also pointed to similar cases observed in other provinces, where power asymmetries between actors with contrasting interests hindered participatory process. Thus, in this case, and according to these authors, insufficient knowledge was not the reason why social-ecological science failed to be incorporated into environmental policy. Rather, it represents the outcome of a wider interplay of socio-political factors.

Based on these cases, it would therefore be possible to anticipate the likelihood of a piece of scientific knowledge to influence environmental policy design and implementation, if - unlike the case described for Caribbean and partly the case in Brazil - it is granted that scientific evidence is solid and relevant to the issue at hand (credibility, legitimacy and relevance), and available in a friendly format to the stakeholders involved. Four pre-requisites, according to Cáceres et al. (2016) are: the engagement of the sectors of society that are likely to benefit or lose, the ability to communicate compelling narratives, the integration with wider social-actor networks, and the emergence of sociopolitical windows of opportunity.

Finally, based on the case of the Arctic, Fleming and Pyenson (2017) argue that the publication of policy-relevant findings in scientific journals is not enough to inform policy. Thus, they suggest that Arctic scientists must directly engage in policy review and revision.

6.7 Urgent issues and emerging solutions

6.7.1 Future scenarios

Recent modelling exercises project future scenarios for the Americas that provide an indication of some policy needs for the region, be it in terms of design, implementation or evaluation. Collectively, such modelling exercises indicate that policy needs are often related to the necessity to reduce greenhouse gas emissions and vulnerability to climate change, to conserve or restore biodiversity and ecosystem services, and to the synergies between these two demands, including the effects on socioeconomic trends. For instance, Chapter 5 shows the results of the GLOBIO model for the Americas, which aims to facilitate the development of policies and strategies to achieve conservation targets and sustainable use of natural resources. GLOBIO employs mean abundance of original species relative to their abundance in undisturbed ecosystems (%) and natural areas (km^2) as indicators of biodiversity (Alkemade et al., 2009). Mean abundance of original species can be interpreted as an indicator for intactness of ecosystems. It shows results for four scenarios (Business as usual or baseline; Global technology; Decentralized solutions; and Consumption change). For the Americas, the Business as usual scenario will produce the highest decrease in mean abundance of original species and there are no substantial differences between the remaining scenarios. However, a small increment in mean abundance of original species could be expected for the Decentralized Solutions scenario in 2050 in South America, Central America and the Caribbean. This scenario relies on local and regional efforts to ensure a sustainable quality of life from a "bottom-up" managed system where small-scale and decentralised technologies are prioritised. On the other hand, an increase in natural areas is probable under Consumption Change pathways for Mexico and South America. Results from GLOBIO show that crop and livestock are the most important drivers in mean abundance of original species and natural areas reduction. It is important to emphasize that GLOBIO does not consider telecoupling processes (see also 6.4.2.3 and Glossary). It is expected that new policy solutions will emerge when telecoupling processes are better understood and incorporated in modelling exercises (Chapter 5).

Based on the sensitivity of different ecosystems to climatic variations in the past 14 years, Seddon et al. (2016) argue that the Arctic tundra (Canada), parts of the boreal forest belt (USA and Canada), the tropical rainforest (especially the Amazon), high montane regions (in the USA, Central America and the Andes), prairies and steppe of North and South America, and the Caatinga deciduous forest in Brazil are probably the most sensitive to climate change in the Americas. Vulnerability is also true for the Caribbean islands, in issues such as sea level rise, natural disasters, water security and biodiversity conservation (Nurse et al., 2014).

All these results to a large extent match the study of Segan et al. (2016) that have modelled the interaction of climate change with land use change and, aiming at reduced climate vulnerability, recommended priorities for conservation and restoration in the Americas and elsewhere. While most of North America (including Mexico), Amazonia, the Andes and the Southern Cone are conservation priorities, the prairies in the USA, Central America and the Caribbean, the Brazilian biodiversity hotspots (Atlantic forest and Cerrado) and the southern grasslands ("pampas") are restoration priorities.

These results are also aligned with the findings of Leadley et al. (2014) that investigated the potential impacts of regime shifts on human-environment systems in the Andes, Amazonia, Cerrado and Caatinga. Their study suggests that moderate to high rates of land-use change at regional scales could act synergistically with high levels of global climate change to cause severe habitat degradation or even habitat loss in terrestrial and freshwater systems. In addition, these regime shifts could have large negative effects on a wide range of ecosystem services. Policies that reduce the possibility of regional-scale regime shifts occurring in central South America will have to deal with local land use change, freshwater management and global climate change, including conservation and restoration. At the regional scale, it would be necessary to reduce the conversion of humid tropical forests and other pristine ecosystems to croplands and pastures and to limit the use of fire. At the global scale, one important challenge is to mitigate climate change without increasing pressure on land use for bioenergy.

6.7.2 Urgent issues

Climate and land-use change, biodiversity and ecosystem loss, and persistence of poverty and inequality are all urgent issues to be addressed in the region, as this Chapter demonstrates. However, in many cases it can be hard to foresee whether existing policies and governance schemes will drive or halt some of the future scenarios described in 6.7.1 (see also Chapter 5). In addition to the challenge related to the fact demonstrated in this Chapter that policies are often designed sectorally (e.g. 6.2), policy evaluation remains a significant a gap.

In some countries of the region this can be at least partly related to insufficiency of monitoring programs that track ecosystem dynamics, their relations with ecosystem services, and human wellbeing. For that purpose, monitoring programs of biodiversity and ecosystem services need to be extended beyond conservation areas. Coordinated monitoring programs are emerging in the region to obtain data on biodiversity and ecosystem services. There are internationally coordinated monitoring programs, such as GLORIA for high mountain biodiversity in the context of climate change, which includes site-based networks in both North America (Millar & Fagre, 2007) and South America (Cuesta et al., 2017). The installation of the basic GLORIA permanent plot settings for vascular plants also stimulated further monitoring approaches on both continents, such as on different animal groups or on socio-economic aspects in the studied regions (Pauli et al. 2015).

Many programs are also arising at the national scale. For example, the FAO is conducting a program focused on helping countries in the region with developing national forest monitoring systems and assessments: Belize, Brazil, Costa Rica, Dominican Republic, Ecuador, El Salvador, Guatemala, Honduras, Nicaragua, Panama, Peru, and Uruguay are part of this program (http://www.fao.org/forestry/fma/73410/en/). On the other hand, since 2004, the forest and soils inventories were established in Mexico. In this inventory, 152 tree and soil variables are obtained every five years, in more than 26,000 sites that cover 57 types of vegetation (http://www.cnf.gob.mx:8090/snif/portal/infys). In addition, given the need of a more reliable monitoring program, this effort expanded and thus the National Biodiversity and Ecosystem Degradation Monitoring System (García-Alaníz et al., 2017) was created in 2016. This monitoring system is a multi-institutional project that collects information on vegetation and wildlife using different sources of information (camera traps, sound recorders, remote sensing and fieldwork) and analyses large amounts of data through machine learning techniques.

New opportunities open up to conduct monitoring programs and use large datasets thanks to: the technological advances in remote sensing; the innovation of in situ data collection (camera traps, acoustic recording, drones among others); the continuous increase in observations made through citizen science; the development of algorithms to process large amounts of data; and more user-friendly platforms to display these data (Stephenson et al., 2017). Because of regional and local socioeconomic differences, the benefits of all these technologies and social initiatives would not be experienced equally in the whole region. This will also depend, for example, on the feasibility to access sites for monitoring, given the natural and safety conditions or social circumstances. A common challenge for the Americas is to find ways to process and translate the generated data into useful and timely information for decision makers. Strategies to merge and interpret all the obtained data will be necessary. In Mexico, field and remote sensing data are integrated in an index that evaluates the integrity of terrestrial ecosystems (Equihua et al., 2014). Maps of this ecosystem integrity index are produced annually and wall-to- wall at a resolution of 1 km^2. The measurement of ecosystem integrity allows an integrated assessment of ecosystems and their capacity to provide ecosystem services and it is planned to guide public policy, providing a platform for evaluating anthropogenic effects on biodiversity and ecosystem services.

6.7.3 Emerging solutions

To the urgent and often integrated issues related to climate and land-use change, biodiversity and ecosystem loss, and persistence of poverty and inequality, this Chapter uncovers some emerging solution. For instance, as seen in 6.6.3, ecosystem-based approaches to climate change adaptation and disaster risk reduction are a great opportunity for the region. Such policies combine biodiversity conservation with climate change mitigation and improvement of livelihoods. For instance, Jantz et al. (2014) demonstrated that it is possible to obtain large benefits in terms of carbon storage and biodiversity conservation if carbon funds are directed at corridors that link existing protected areas (see also Venter, 2014). Preserving corridors between protected areas could maintain habitat connectivity across landscapes, mitigate the effects of land use and climate change on biodiversity, and improve livelihoods.

The USA has a number of existing policies that potentially address landscape connectivity and permeability (Kostyack et al., 2011). Many of the existing networks (local, national, and international; see 6.3.1, 6.3.2) provide an opportunity for sharing both information and solutions, which facilitate mainstreaming and scalability of ecosystem-based approaches. Whereas adaptation policies primarily address vulnerability and risks (see 6.6.3), sustainable development policies aim to reduce poverty via economic growth, address inequality via redistribution of wealth, and prevent environmental degradation by using resources sustainably (Agrawal & Lemos, 2015). However, whenever adaptation avoids or reduces climate risks without negatively impacting human systems and natural systems, it becomes an important subset of the sustainable development agenda (Juhola et al., 2016; Pant et al., 2015; Kasecker et al., 2017).

Therefore, policies and actions that reduce poverty while protecting and/or restoring ecosystems are potentially adaptive to climate change, particularly in developing countries (Scarano, 2017). In parallel, actions that enable sustainable development locally or nationally can accelerate successful climate change adaptation globally (IPCC, 2014). Adaptation is an important step in the transition to sustainability, and such actions require capacity, investment, integrated policies and adequate governance – all of which require participation and dialogue.

As seen throughout this Chapter, the region has many local solutions emerging (e.g. Table 6.1), but still many national and regional contradictions in this respect (e.g. Figure 6.1). Perhaps even more relevant is the notion that sustainable development is one among other existing options. Other options emerge in the region from distinct cosmovisions, such as in the case of buen vivir (see 6.4.3.2), or as a rejection of economic growth as the only alternative, in the case of degrowth (Kothari et al., 2014). Although scientific output on degrowth is largely European, USA and Canada also have a relevant contribution to this line of thought and, as a movement, it has presence in other countries in the region such as Colombia and Cuba as well (Weiss & Cattaneo, 2017). Beling et al. (2018) suggest that synergies between sustainable development, buen vivir and degrowth can compensate for the caveats of each discourse and “open pathways towards a global new Great Transformation”.

6.8 Conclusions

This Chapter concludes that for most countries of the region, environmental and development policies are often conceived, designed and implemented separately, from a sectoral viewpoint (see 6.2). Since development pressures frequently outpace or outweigh environmental policies, the development process becomes unsustainable and a key driver of biodiversity and ecosystem services loss (6.1.1). This is especially true for the developing countries in the Americas region, and accounts for many of the negative trends in biodiversity and ecosystem services that are evident across the region. On the positive side, however, the region still harbors astonishing biodiversity and ecosystem services of local and global importance related to water, climate and food security (6.1.1). Moreover, there are reported reductions in rate of habitat loss in specific biomes. Although success in this respect is often times more local than national or subregional, it can be at least partly attributed to a broad array of policy instruments, which include regulatory (e.g. protected areas) and incentive mechanisms (e.g. eco-certification, financial incentives, offsets), as well as those originated from diverse views of the relationship between man and nature (e.g. management of the system of life applied in Bolivia, based in rights, duties and obligations) (6.4; and Table 6.5). A diversity of cases across policy areas, levels of economic development, and political cultures suggest that partnerships and participatory deliberative processes contribute to a large class of problem-solving situations and can support successful governance (6.3.1), and this Chapter also uncovers a number of examples of such good practices (e.g. Table 6.1).

Despite some of these good news, the net biodiversity and ecosystem services loss that is currently evident in almost every aspect of the region’s natural ecosystems is expected to continue through to 2050, if society does not change business-as-usual patterns of land-use change and greenhouse gas emissions (6.7.1). This will result in reductions in the adaptive capacity of the societies throughout the region, especially poor communities in Latin America and the Caribbean (6.6.3; 6.7.3). There is a great regional opportunity to incorporate ecosystem-based strategies into national and sub-national-level development planning and this can be done sooner than later (6.6.3; 6.7.3), especially if one considers that the cost of recovery of ecosystems and species is high (6.4.1.2). Although there is much optimism and some evidence of the potential of habitat restoration in the region, this is often costly (6.4.1.2). This indicates that countries are likely to benefit from acting quickly to invest in the preservation and sustainable use of their existing ecological infrastructure.

There is an overall gap of policy evaluation in the Americas, which is more pronounced in Latin America and the Caribbean than it is in North America (6.7.2). Information on policy effectiveness is often derived through case studies and anecdotal accounts. Evaluation could benefit from improved monitoring systems, involving both new technologies and community-based monitoring at local level. It could also benefit from improved analytical tools that integrate biodiversity and ecosystem services variables and human and socioeconomic development variables (6.7.2). For instance, although there begin to emerge evidences of leakage and spillover effects in many levels and scales across the region, they remain understudied. Cases where environmentally damaging activities are relocated elsewhere after being stopped locally are found from protected area level to biome level. Leakage and spillover effects are often unforeseen either due to lack of systemic planning or adequate mapping of potential stakeholders (6.3.4; 6.4.2.3).

The sociocultural diversity of the region is also an untapped opportunity. Indigenous peoples throughout the Americas have developed many different socioecological and governance systems (nationally and locally), which exist in parallel to mainstream governance (6.3.5). Although conflicts persist both in countries that acknowledge such rights and countries that do not, indigenous and local knowledge and practices can positively influence biodiversity and ecosystem services (6.3.1; 6.3.5; 6.4.1.1; 6.4.3.2; 6.6.1).

For most countries in the region, global commitments (SDG, Aichi targets, Paris Accord) are often uncoupled from national policies. As a result, the rate of achievement of global commitments vary largely between countries. Thus, bringing global commitments down to local level implementation, and scaling up local solutions to global diplomacy remains challenging (6.5). Some of the difficulties might be related to issues such as the possible confusion created by excessive list of targets to be achieved and indicators to measure them (Easterly, 2015; Lomborg, 2017), or to science gaps around key concepts such as sustainability or planetary boundaries, for instance (Montoya et al., 2018). Understanding synergies and trade-offs between goals within specific global agreements (e.g. Di Marco et al., 2015; Pradhan et al., 2017) and between distinct global agreements (e.g. Von Stechow et al., 2016; Le Gouvello et al., 2017) can be an important step to reduce confusion and enhance focus for policy design and implementation at national and local level.

Table 6.5. Policy options in the Americas: instruments, enabling factors and country-level challenges. Chapter sections cited include those in this Chapter 6, but also sections of the other Chapters in this report. SU=sustainable use; RE = recovery or rehabilitation of natural and/or human systems; PR = protection.

Table 6 5 Examples of policy options in the Americas: instruments, enabling factors and country-level challenges.

SU=sustainable use; RE = recovery or rehabilitation of natural and/or human systems; PR = protection.

POLICY INSTRUMENTS	GOALS			ENABLING FACTORS (Way forward)	IMPEDIMENTS (Challenges more common to some countries than others)	CHAPTER -SECTION
	SU	RE	PR			
1. REGULATORY MECHANISMS						6 – 6.4.1
1.1 AREA-BASED						-
Protected areas	√	√	√	Legal basis for protecting or setting aside specific areas Community support for exclusionary measures Effective management authority by State, community or private sector Adequate resources for monitoring and enforcement	Weak or unstable legal basis for multi-sectoral management measures Insecure funding for on-going surveillance and enforcement of protection measures Low compliance with protection measures Lack of community support for measures Private sector investments threatened by spatial exclusions Fragmentation of sites and/or inadequate spatial connectivity	3 – 3.5.2 6 – 6.4.1.1
Other effective area-based conservation measures (OECM) (e.g., set-asides[1])	√	√	√			2 – Box 2.4 2 – 2.3.2 2 – 2.3.5 3 – Box 3.1 3 – 3.3.4 3 – 6 4 – Box 4.5 5 – 5.4.7 5 – 5.4.10 6 – 6.4.1.1
Indigenous and Community Conserved Areas (ICCA)	√	√	√	Capacity of self-organization Official acknowledgement of rights consistent with national legislation Mechanisms allowing co-management and/or self-governance systems	Weak or missing recognition of indigenous peoples and local communities rights and ownership/access to land by Central governments, neighboring communities or private sector	2 – 2.2.6 3 – 3.4.1.1 5 – 5.4.11 6 – 6.4.1.1 6 – 6.4.1.2
1.2 LIMITS						-
To technology (e.g., pollution control)	√		√	Adequate background information and risk analysis to set limits Technological advances to reduce or mitigate pollution /by-products while maintaining economic efficiency Adequate resources for monitoring and enforcement	Disproportionate political influence of industries Technological advances that outstrip or negate control mechanisms Low risk aversion in setting limits Weak monitoring and surveillance for compliance	3 – 3.2.2.3 3 – 3.2.3.2 3 – 3.2.4 4 – 4.4.2 6 – 6.2.1 6 – 6.6.2
To access (e.g., tourism, fisheries)	√		√	Governance capacity at local level Clear rules to manage potential sources of revenue Social cohesion and participation	Inability to regulate access to areas Lack of human and financial resources Excessive expectations from the market of enhanced consumer demand Inadequate sharing of benefits	4 – Box 4.19 4 – 4.3.3 6 – 6.6.1
1.3 MANAGEMENT						-
Ecosystem restoration	√	√		Technological and knowledge availability Economic incentives to overcome high costs favourable policy environment to promote restoration Funding for up-front costs to undertake restoration Mechanisms for cost recovery of benefits from successes	Lack of recognition of restoration in legal frameworks Inadequate funding for continuity of initiatives Insufficient knowledge to design effective restoration strategies for specific sites Lack of elimination of causes of original degradation Unreal expectations of time or funding needed for restoration to reach goals	2 – 2.2.8 2 – 2.2.11 2 – 2.2.13 4 – 4.4.1 5 – 5.4.7 6 – 6.4.1.2
Ecosystem-based approaches (e.g., EbA[2] and EcoDRR[3])	√	√	√	Availability of financing Receptiveness of industries to take on additional operating costs Inclusive governance with policy endorsement of Ecosystem Approaches to Management (use of the best knowledge available)	Weaknesses in science basis for broadening management context and accountabilities Lack of cost-effective operational tools to address full ecosystem effects of sectoral actions Lack of knowledge of transferability of progress from project to project Absence of policy framework explicitly calling for ecosystem approaches at sectoral levels	3 – 3.6 4 – Box 4.14 4 – 4.4.3 4 – 4-4.5 6 – 6.6.3

POLICY INSTRUMENTS	GOALS			ENABLING FACTORS (Way forward)	IMPEDIMENTS (Challenges more common to some countries than others)	CHAPTER -SECTION
	SU	RE	PR			
Control of Invasive-Alien Species (IAS)	√	√	√	Strong regulatory frameworks for pathways of introductions Availability of technologies for management and control Adequate monitoring for early detection Local capacity and collaboration networks for site-level mobilization of community resources for management or elimination	Shortage of scientific information on invasion pathways and likelihood of successful establishment Low awareness of risks by people involved in major invasion pathways Inadequate facilities for interception and quarantine facilities Inadequate or insecure funding for ongoing interception, monitoring and control	2 – 2.2.15 2 – 2.3.4 3 – 3.2.2.3 3 – 3.2.3.2 3 – 3.2.4.2 3 – 6 4 – 4.4.4 6 – Box 6.3
2. INCENTIVE MECHANISMS						6 – 6.4.3
Payment for Ecosystem Services (PES)	√	√	√	Trust building between service users and providers Direct linkages between buyers and sellers Adequate metrics for calculating payments Fair and transparent markets for exchange of payments Adequate monitoring when payment is for ongoing provision of services	Low return on investment for those paying for services Weak information basis for calculating appropriate payments Land tenure rights not adequate protected from payment arrangements Power structures that do not promote equitable and transparent payment agreements or distribution of payments Lack of recognition of non-market values of Nature and NCP when negotiating payment agreements, or lack of measures or governance processes to protect to values	2 – 2.5.1 4 – 4.3.1 6 – 6.4.2.1
Offsets	√	√		Sufficient science / knowledge base to quantify both impacts and expected benefits form offsets; Sufficient legal basis to authorize offsets as a mitigation options Adequate capacity for enforcement management and monitoring; Transparent and inclusive settings for establishing appropriate trade-offs of offsets for likely impacts.	Many weaknesses or gaps in knowledge basis for trade-off metrics, establishing equivalence, additionality, reversibility and appropriate time-scales, longevity Low availability of areas for spatial delivery of offsets Lack of resources for ongoing compliance monitoring Low adaptability of agreements on offsets, once established, if monitoring shows that benefits accruing are lower than expected or impact higher	6 – 6.4.2.2
Eco-certification	√			Adequate knowledge to set and enforce standards Reliable chain of custody for certified products Demand in high-value markets that can bear price increment for certainty of sustainability, High consumer recognition and credibility for certification labels	Weak government – private sector linkages High up-front costs to demonstrate sustainable practices and earn certification, before any economic benefits are realized Increases in operating costs so large that market competitiveness may be lost Lack of transparency in markets	2 – 2.2.1.3 2 – 2.2.1.5 2 – 2.2.2.1 6 – 6.4.2.3
3. RIGHTS-BASED APPROACHES						6 – 6.4.2
Rights of Mother Earth	√		√	Capacity of self-organization Official acknowledgement of rights consistent with national legislation Mechanisms allowing co-management and/or self-governance systems	Inadequate recognition of "rights" of Non-human persons in law Challenges in delimiting when such rights would be transgressed in areas already urbanized or under intensive cultivation	2 – 2.4 3 – Box 3.3 4 – Box 4.7 6 – 6.3.5
Access and Benefit Sharing (ABS)	√			Human and institutional capacities to grant access Capacity to monitor and negotiate mutually agreed terms Robust legal frameworks to require sharing benefits Inclusive, participatory mechanisms for establishing agreements	Weak legal basis to require benefit sharing of many uses of Nature Unrealistic expectations of quantity of monetary benefits Complexity and lengthy procedures for setting benefits Fundamental challenges to property rights, including intellectual property rights	2 – 2.4 2 – 2.5 2 – Box 2.6 2 – 2.7 6 – 6.4.2.4

1. Set-asides: areas set-aside for conservation inside private properties; 2. EbA = ecosystem-based adaptation to climate change; 3. EcoDRR = ecosystem-based disaster risk reduction.

Source: Own representation

6.9 References

ACTO (2014). *Work Plan 2014.* Amazon Cooperation Treaty Organization, 22 p. Available at: http://otca.info/portal/admin/_upload/plano_trabalho/201-WORK.PLAN_2014.pdf

Adams, W. M., & Hutton, J. (2007). People, parks and poverty: political ecology and biodiversity conservation. *Conservation and Society*, 5, 147-183.

Adger, W. N., Brown, K., & Tompkins, E.L. (2006). The political economy of cross-scale networks in resource co-management. *Ecology and Society*, 10(2), 9

Agardy, T., di Sciara, G. N., & Christie, P. (2011). Mind the gap: addressing the shortcomings of marine protected areas through large scale marine spatial planning. *Marine Policy*, 35, 226-232.

Agrawal, A., & Lemos, M.C. (2015). Adaptive development. *Nature Climate Change*, 5, 186–187.

Alexander, S. M., Andrachuk, M., & Armitage, D. (2016). Navigating governance networks for community-based conservation. *Frontiers in Ecology and the Environment*, 14(3), 155-164.

Alkemade, R., Van Oorschot, M., Miles, L., Nellemann, C., Bakkenes, M., & Ten Brink, B. (2009). GLOBIO3: A framework to investigate options for reducing global terrestrial biodiversity loss. *Ecosystems*, 12, 374–390.

Anderson, M. B., Hall, D. M., McEvoy, J., Gilbertz, S. J., Ward, L., & Rode, A. (2016). Defending dissensus: participatory governance and the politics of water measurement in Montana's Yellowstone River Basin. *Environmental Politics*, https://doi.org/10.1080/09644016.2016.1189237

Andrade, A. B. de, & Soares, M. O. (2017). Offshore marine protected areas: Divergent perceptions of divers and artisanal fishers. *Marine Policy*, 76, 107–113.

Anthamatten, P., & Hazen, H. (2014). Changes in the global distribution of protected areas, 2003–2012. *The Professional Geographer*, 67(2), 195-203.

Arantes, M. L., & Freitas, C. E. C. (2016). Effects of fisheries zoning and environmental characteristics on population parameters of the tambaqui (*Colossoma macropomum*) in managed floodplain lakes in the Central Amazon. *Fisheries Management and Ecology,* 23(2), 133-143.

Armitage, D. R. (2005). Adaptive capacity and community-based natural resources management. *Environmental Management*, 35(6), 703-715.

Armitage, D. R., Plummer, R., Berkes, F., Arthur, R. I., Charles, A. T., Davidson-Hunt, I. J,. Diduck, A. P., Doubleday, N. C., Johnson, D. S., Marschke, M., McConney, P., Pinkerton, E. W., & Wollenberg, E. K. (2009). Adaptive co-management for social–ecological complexity. *Frontiers in Ecology and the Environment*, 7(2), 95–102.

Aronson, J., Blignaut, J. N., Milton, S. J., Le Maitre, D., Esler, K. J., Limouzin, A., Fontaine, C., de Wit, M. P., Mugido, W., Prinsloo, P., van der Elst, L., & Lederer, N. (2010). Are socioeconomic benefits of restoration adequately quantified? A meta-analysis of recent papers (2000–2008) in Restoration Ecology and 12 other scientific journals. *Restoration Ecology*, 18 (2), 143–154.

Aronson, J., Brancalion, P. H. S., Durigan, G., Rodrigues, R. R., Engel, V. L., Tabarelli, M., Torezan, J. M. D., Gandolfi, S., de Melo, A. C. G., Kageyama, P. Y., Marques, M. C. M., Nave, A. G., Martins, S. V., Gandara, F. B., Reis, A., Barbosa, L. M., & Scarano, F. R. (2011). What role should government regulation play in ecological restoration: ongoing debate in São Paulo State, Brazil. *Restoration Ecology*, 19, 690–695.

Ascher, W. (2007). Policy sciences contributions to analysis to promote sustainability. *Sustainability Science*, 2, 141-149.

Asquith, N. M., Vargas, M. T., & Wunder, S. (2008). Selling two environmental services: In-kind payments for bird habitat and watershed protection in Los Negros, Bolivia. *Ecological Economics*, 65, 675-684.

Azadi, H., & Ho, P. (2010). Genetically modified and organic crops in developing countries: a review of options for food security. *Biotechnology Advances*, 28, 160–168.

Azevedo-Santos, V. M., Fearnside, P. M., Oliveira, C. S., Padial, A. A., Pelicice, F. M., Lima Jr, D. P., Simberloff, D., Lovejoy, T. E., Magalhães, A. L. B., Orsi, M. L., Agostinho, A. A., Esteves, F. A., Pompeu, P. S., Laurance, W. F., Petrere Jr, M., Mormul, R. P., & Vitule, J. R. S. (2017). Removing the abyss between conservation science and policy decisions in Brazil. *Biodiversity Conservation*, 26(7), 1745-1752. https://doi.org/10.1007/s10531-017-1316-x.

Baker, S., Eckerberg, K., & Zachrisson, A. (2013). Political science and ecological restoration. *Environmental Politics*, 23(3), 509-524.

Ballari, S. A., Anderson, C. B., & Valenzuela, E. J. (2016). Understanding trends in biological invasions by introduced mammals in southern South America: a review of research and management. *Mammal Review*, 46(3), 229-240. https://doi.org/10.1111/mam.12065.

Balmford, A., Green, J. M. H., Anderson, M., Beresford, J., Huang, C., Naidoo, R., Walpole, M., & Manica, A. (2015). Walk on the wild side: estimating the global magnitude of visits to protected areas. *PLoS Biology*, 13(2), e1002074. https://doi.org/10.1371/journal.pbio.1002074

Balvanera, P., Uriarte, M., Almeida-Leñero, L., Altesor, A., DeClerck, F., Gardner, T., Hall, J., Lara, A., Laterra, P., Peña-Claros, M., Silva Matos, D. M., Vogl, A. L., Romero-Duque, L. P., Arreola L. F., Caro-Borrero, Á. P., Gallego, F., Jain, M., Little, C., de Oliveira Xavier, R., Paruelo, J. M., Peinado, J. E., Poorter, L., Ascarrunz, N., Correa, F., Cunha-Santino, M. B., Hernández-Sánchez, A. P., & Vallejos, M. (2012): Ecosystem services research in Latin America: the state of the art. *Ecosystem Services*, 2, 56-70.

Ban, N. C., Davies, T., Aguilera, S. E., Brooks, C., Cox, M., Epstein, G., Evans, L. S., Maxwell, S. M., & Nenadovic, M. (2017). Social and ecological effectiveness of large marine protected areas. *Global Environmental Change*, 43, 82–91.

Barnett, J., Tschakert, P., Head, L., & Adger, W. N. (2016) A science of loss. *Nature Climate Change*, 6(11), 976-978.

Barral, P., Rey Benayas, J. M., Meli P., & Maceira, N. (2015): Quantifying the impacts of ecological restoration on biodiversity and ecosystem services in agroecosystems: a global meta-analysis. *Agriculture, Ecosystem and Environment*, 202, 223–231.

Bartelmus, P. (2015). Do we need ecosystem accounts? *Ecological Economics*, 118, 292-298.

Basurto, X., Gelcich, S., & Ostrom, E. (2013). The socio-ecological system framework as a knowledge classification system for benthic small-scale fisheries. *Global Environmental Change*, 23(6), 1366-1380.

BBOP (2012). *Resource Paper: No net loss and loss–gain calculations in biodiversity offsets.* Business and Biodiversity Offsets Programme, Washington, DC.

Begossi, A., May, P. H., Lopes, P. F., Oliveira, L. E. C., da Vinha, V., & Silvano, R. A. M. (2011). Compensation for environmental services from artisanal fisheries in SE Brazil: policy and technical strategies. *Ecological Economics*, 71, 25-32.

Beling, A. E., Vanhulst, J., Demaria, F., Rabi, V., Carballo, A. E., & Pelenc, J. (2018). Discursive synergies for a 'Great Transformation' towards sustainability: Pragmatic contributions to a necessary dialogue between Human Development, Degrowth, and Buen Vivir. *Ecological Economics*, 144, 304-313.

Bennett, N. J., Blythe, J., Tyler, S., & Ban, N. C. (2015). Communities and change in the Anthropocene: understanding social-ecological vulnerability and planning adaptations to multiple interacting exposures. *Regional Environmental Change*, 16(4), 907-926.

Berkes, F. (2006). From community-based resource management to complex systems: the scale issue and marine commons. *Ecology and Society*, 11(1), 45-56.

Berkes, F. (2009). Evolution of co-management: Role of knowledge generation, bridging organizations and social learning. *Journal of Environmental Management*, 90, 1692–1702.

Biggs, R., Schlüter, M., Biggs, D., Bohensky, E. L., BurnSilver, S., Cundill, G., Dakos, V., Daw, T. M., Evans, L. S., Kotschy, K., Leitch, A. M., Meek, C., Quinlan, A., Raudsepp-Hearne, C., Robards, M. D., Schoon, M. L., Schultz, L., & West, P. C. (2012). Toward principles for enhancing the resilience of ecosystem services. *Annual Review of Environmental Resources*, 37, 421–48.

Bishop, J. (Ed.), (2012). *TEEB – The economics of ecosystem and biodiversity in business and enterprise.* Earthscan, Routledge, London.

Blackman, A., & Naranjo, M. A. (2012). Does eco-certification have environmental benefits? Organic coffee in Costa Rica. *Ecological Economics*, 83, 58–66.

Blackman, A., & Rivera, J. (2011). Producer-level benefits of sustainability certification. *Conservation Biology*, 25(6), 1176–1185.

Boden, T. A., Marland, G., & Andres, R. J. (2015). *National CO_2 Emissions from Fossil-Fuel Burning, Cement Manufacture, and Gas Flaring: 1751-2011*, Carbon Dioxide Information Analysis Center, Oak Ridge National Laboratory, U.S. Department of Energy. https://doi.org/10.3334/CDIAC/00001_V2015.

Borges Hernández, T., Coya de la Fuente, L., & Wald, K. L. (2008). Protected areas and tourism in Cuba. In A. Stronza & W. H. Durham (Eds.), *Ecotourism and Conservation in the Americas* (pp. 179-90). Wallingford, UK: Cabi International.

Börner, J., Baylis, K., Corbera, E., Ezzine-de-Blas, D., Ferraro, P., Honey-Rosés, J., Lapeyre, R., Persson, M., & Wunder, S. (2016). Emerging evidence on the effectiveness of tropical forest conservation. *PLoS One*, 11(11), e0159152

Borrini-Feyerabend, G., Kothari, A., & Oviedo, G. (2004). *Indigenous and Local Communities and Protected Areas - Towards Equity and Enhanced Conservation: Guidance on policy and practice for co-managed areas and Community Conserved Areas.* (Best Practice Protected Area Guidelines Series n.11 - Adrian Phillips, Series Editor). World Commission on Protected Areas (WCPA). IUCN. (pp. 139)

Bourne, A., Holness, S., Holden, P. et al., (2016). A socio-ecological approach for identifying and contextualising spatial ecosystem-based adaptation priorities at the sub-national level. *PLoS One*, 11(5), e0155235. https://doi.org/10.1371/journal.pone.0155235

Boyd, J. (2002). Financial responsibility for environmental obligations: Are bonding and assurance rules fulfilling their promises? *Research in Law and Economics*, 20, 417–486.

Brancalion, P. H. S., Garcia, L. C., Loyola, R., Rodrigues, R. R., Pillar, V. D., & Lewinsohn, T. M. (2016). A critical analysis of the Native Vegetation Protection Law of Brazil (2012): updates and ongoing initiatives. *Natureza e Conservação*, 14(S1), 1-15.

Braza, M. (2017). Effectiveness of conservation easements in agricultural regions. *Conservation Biology*, 31, 848-859.

Brink, E., Aalders, T., Ádám, D., Feller, R., Henselek, Y., Hoffmann, A., Ibe, K., Matthey-Doret, A., Meyer, M., Negrut, N. L., Rau, A.-L., Riewerts, B., von Schuckmann, L., Törnros, S., von Wehrden, H., Abson, D. J., & Wamsler, C. (2016). Cascades of green: A review of ecosystem-based adaptation in urban areas. *Global Environmental Change*, 36, 111-123.

Brito, D., Ambal, R. G., Brooks, T., De Silva, N., Foster, M., Hao, W., Hilton-Taylor, C., Paglia, A., Rodríguez, J. P., & Rodríguez, J. V. (2010). How similar are national red lists and the IUCN Red List? *Biological Conservation*, 143(5), 1154-1158.

Brondizio, E.S., Ostrom, E., & Young, O. R. (2009). Connectivity and the governance of multilevel social-ecological systems: the role of social capital. *Annual Review of Environmental Resources*, 34, 253-278.

Brook, B. W., Ellis, E. C., Perring, M. P., Mackay, A. W., & Blomqvist, L. (2013). Does the terrestrial biosphere have planetary tipping points? *Trends in Ecology and Evolution*, 28, 396-401.

Brownlie, S., King, N., & Treweek, J. (2013). Biodiversity tradeoffs and offsets in impact assessment and decision making: can we stop the loss? *Impact Assessment and Project Appraisal,* 31(1), 24–33.

Bruner, A. G., Gullison, R. E., Rice, R. E., & da Fonseca, G. A. B. (2001). Effectiveness of parks in protecting biodiversity. *Science*, 291, 125-128.

Buckley, R., (2009). Evaluating the net effects of ecotourism on the environment: a framework, first assessment and future research. *Journal of Sustainable Tourism*, 17(6), 643–672.

Bullock, J. M., Aronson, J., Newton, A. C., Pywell, R. F., & Rey-Benayas, J. M. (2011). Restoration of ecosystem services and biodiversity: conflicts and opportunities. *Trends in Ecology and Evolution*, 23(10), 541-549.

Burgeff, C., Huerta, E., Acevedo, F., & Sarukhán, J. (2014). How much can GMO and non-GMO cultivars coexist in a megadiverse country? *AgBioForum*, 17(1), 90-101.

Burney, J., Cesano, D., Russell, J., La Rovere, E. L., Corral, T., Coelho, N. S., & Santos, L. (2014). Climate change adaptation strategies for smallholder farmers in the Brazilian Sertão. *Climatic Change*, 126, 45–59.

Butler, W.H., & Goldstein, B. E. (2010). The US Fire Learning Network: Springing a Rigidity Trap through Multiscalar Collaborative Networks. *Ecology and Society*, 15(3), 21. http://www. ecologyandsociety.org/vol15/iss3/art21/

Cabrera Medaglia, J., (2013). *The implementation of the Nagoya Protocol in Latin America and the Caribbean: Challenges and Opportunities.* In E. Morgera, M. Buck, & E. Tsioumani (Eds.), *The 2010 Nagoya Protocol on Access and Benefit-Sharing in Perspective - Implications for International Law and Implementation Challenges* (pp. 331-368). Leiden, Netherlands, Boston, United States: Martinus Nijhoff,

Cáceres, D. M., Silvetti, F., & Díaz, S. (2016). The rocky path from policy-relevant science to policy implementation — a case study from the South American Chaco. *Current Opinion in Environmental Sustainability*, 19, 57–66.

Campos, A. G., & Chaves, J. V. (2014). *Seguro defeso: Diagnóstico dos problemas enfrentados pelo programa, Texto para Discussão*, Instituto de Pesquisa Econômica Aplicada (IPEA), No. 1956, Brasília, Brasil.

Canada (2014). *Canada´s Fifth National Report to the Convention on Biological Diversity.* Canada Wildlife Service, Environment Canada, 112 p. Available at: https://www.cbd.int/doc/ world/ca/ca-nr-05-en.pdf

Canada (2016). *2020 Biodiversity goals and targets for Canada,* 4 p. Available at: http://publications.gc.ca/collections/collection_2016/eccc/CW66-524-2016-eng.pdf

Carneiro da Cunha, M., & Morim de Lima, A. G. (2017) How Amazonian indigenous peoples contribute to biodiversity. In B. Baptiste, D. Pacheco, M. Carneiro da Cunha, & S. Diaz (Eds.), *Knowing our Lands and Resources: Indigenous and Local Knowledge of Biodiversity and Ecosystem Services in the Americas*. Knowledges of Nature 11 (pp. 63-81). UNESCO: Paris.

Cash, D. W., Adger, W. N., Berkes, F., Garden, P., Lebel, L., Olsson, P., Pritchard, L., & Young, O. (2006). Scale and cross-scale dynamics: governance and information in a multilevel world. *Ecology and Society*, 11(2), 8. http://www.ecologyandsociety.org/vol11/iss2/art8/

Castello, L., McGrath, D. & Beck, P. S. A. (2011). Resource sustainability in small-scale fisheries in the Lower Amazon floodplains. *Fisheries Research*, 110, 356-364.

Castello, L., Viana, J. P., Watkins, G., Pinedo-Vasquez, M., & Luzadir, V. A. (2009). Lessons from integrating fishers of Arapaima in small-scale fisheries management at the Mamirauá Reserve, Amazon. *Environmental Management*, 43, 197-209.

Castro, F de, Hogenboom, B., & Baud, M. (2016). Introduction: Environment and Society in Contemporary Latin America. In Castro, F de, B. Hogenboom, & M. Baud (Eds.) *Environmental Governance in Latin America* (pp. 1-25). Hampshire, United Kingdom: Palgrave MacMillan.

CBD (2012). *Climate Change and Biodiversity.* Convention of Biological Diversity. Available at: https://www.cbd.int/climate/intro.shtml

CBD (2016). *Biodiversity and the 2030 Agenda for Sustainable Development. Technical Note.* Convention on Biological Diversity. Available at: https://www.cbd.int/development/doc/ biodiversity-2030-agenda-technical-note-en.pdf. Accessed August 2017.

CBD/UNEP (2010). *Strategic Plan for Biodiversity 2011-2020 and the Aichi Targets "Living in Harmony with Nature"*. Convention of Biological Diversity, United Nations Environment Programme. https://www.cbd.int/sp/default.shtml

CDP (2015a). *Accelerating action CDP Global Water Report 2015.* The Carbon Disclosure Project.

CDP. (2015b). *CDP Global Climate Change Report 2015.* The Carbon Disclosure Project.

Ceccon, E., Barrera-Cataño, J. I., Aronson, J., & Martínez-Garza, C. (2015). The socioecological complexity of ecological restoration in Mexico. *Restoration Ecology*, 23(4), 331-336.

CEPAL (2007). *Políticas sociales. La corrupción y la impunidad en el marco del desarrollo en América Latina y el Caribe: un enfoque centrado en derechos desde la perspectiva de las Naciones Unidas*. CEPAL, Santiago de Chile.

CEPAL (2012). *La inversion extranjera directa em America Latina y Caribe*. CEPAL, Santiago de Chile.

CEPAL (2015). *Planificación y prospectiva para la construcción de futuro en América Latina y el Caribe.* Textos seleccionados 2013-2016.

Chape, S., Harrison, J., Spalding, M., & Lysenko, I. (2005). Measuring the extent and effectiveness of protected areas as an indicator for meeting global biodiversity targets. *Philosophical Transactions of the Royal Society B*, 360, 443–455.

Chaves, M., Macintyre, T., Verschoor, G., & Wals, A. E. J. (2018). Radical ruralities in practice: Negotiating *buen vivir* in a Colombian network of sustainability. *Journal of Rural Studies*, 59, 153-162. https://doi.org/10.1016/j.jrurstud.2017.02.007

Chaves, R. B., Durigan, G., Brancalion, P. H. S., & Aronson, J. (2015). On the need of legal frameworks for assessing restoration projects success: new perspectives from São Paulo state (Brazil). *Restoration Ecology*, 23(6), 754-759.

Chishakwe, N., & Young, T. R. (2003). Access to Genetic Resources, and Sharing the Benefits of their Use: International and Sub-regional Issues. *Informe inédito, IUCN*, Switzerland, 18 pp.

Christian, C., Ainley, D., Bailey, M., Dayton, P., Hocevar, J., LeVine, M., Nikoloyuk, J., Nouvian, C., Velarde, E., Werner, R. & Jacquet, J. (2013). Questionable stewardship: A review of formal objections to MSC fisheries certifications. *Biological Conservation*, 161, 10-17.

Clapp, J. (2015). Distant agricultural landscapes. *Sustainability Science*, 10, 305–316.

Coad L., Leverington, F., Knights, K., Geldmann, J., Eassom, A., Kapos, V., Kingston, N., de Lima, M., Zamora, C., Cuardros, I., Nolte, C., Burgess, N. D., & Hockings, M. (2015). Measuring impact of protected area management interventions: current and future use of the Global Database of Protected Area Management Effectiveness. *Philosophical Transactions of the Royal Society B* 370, 20140281. http://dx.doi.org/10.1098/rstb.2014.0281

Cochrane, K. (2015). Use and misuse of CITES as a management tool for commercially-exploited aquatic species. *Marine Policy*, 59, 16-31.

Coralie, C., Guillaume, O., & Claude, N. (2015). Tracking the origins and development of biodiversity offsetting in academic research and its implications for conservation: a review. *Biological Conservation*, 192, 492-503. https://doi.org/10.1016/j.biocon.2015.08.036.

Coria, J., & Calfucura, E. (2012). Ecotourism and the development of indigenous communities: The good, the bad, and the ugly. *Ecological Economics*, 73, 47–55.

Correa, C. (2005). Do national access regimes promote the use of genetic resources and benefit sharing? *International Journal of Environment and Sustainable Development*, 4(4), 444-463.

Costa, G. D. F, Menger, K. R., & Tancredi, L. (2015). A reformulação dos eixos da IIRSA. *UFRGSMUN - UFRGS Model United Nations*, 3, 135-168.

Costedoat, S., Corbera, E., Ezzine-de-Blas, D., Honey-Rosés, J., Baylis, K., & Castillo-Santiago, M. A. (2015). How effective are biodiversity conservation payments in Mexico? *PLoS One*, 10(3), e0119881. https://doi.org/10.1371/journal.pone.0119881.

Cox, M., Arnold, G., & Tomás, S. V. (2010). A review of design principles for community-based natural resource management. *Ecology and Society*, 15(4), 38. http://www. ecologyandsociety.org/vol15/iss4/art38/

Cronkleton, P., Guariguata, M. R., & Albornoz, M. A. (2012). Multiple use forestry planning: timber and Brazil nut management in the community forests of Northern Bolivia. *Forest Ecology and Management*, 268, 49-56.

Crouzeilles, A. R., Curran, M., Ferreira, M. S., David, B., Grelle, C. E. V, Benayas, J. M. R. (2016). Ecological drivers of forest restoration success: a global meta-analysis. *Nature Communications*, 7, 1–8. https://doi.org/10.1038/ncomms11666

Crowl, T. A., Crist, T. O., Parmenter, R. R., Belovsky, G., & Lugo, A. E. (2008). The spread of invasive species and infectious disease as drivers of ecosystem change. *Frontiers in Ecology and the Environment*, 6(5), 238–246.

Crozier, M. (2008). Listening, learning, steering: new governance, communication and interactive policy formation. *Policy and Politics*, 36(1), 3-9.

Curran, M., Hellweg, S., and Beck, J. (2014). Is there any empirical support for biodiversity offset policy? *Ecological Applications*, 24(4), 617–632.

Danielsen, F., Burgess, N. D., & Balmford, A. (2005). Monitoring matters: examining the potencial of locally-based approaches. *Biodiversity and Conservation*, 14, 2507-2542.

Danielsen, F., Burgess, N. D., Balmford, A., Donald, P. F., Funder, M., Jones, J. P. G., Alviola, P., Balete, D. S., Blomley, T., Brashares, J., Child, B., Enghoff, M., Fjeldsa, J., Holt, S., Hubertz, H., Jensen, A. E., Jensen, P. M., Massao, J., Mendoza, M. M., Ngaga, Y., Poulsen, K., Rueda, R., Sam, M., Skielboe, T., Stuart-Hill, G., Topp-Jorhensen, E., & Yonten, D. (2008). Local participation in natural resource monitoring: a characterization of approaches. *Conservation Biology*, 23(1), 31-42.

Danielsen, F., Jensen, P. M., Burgess, N. D., Altamirano, R., Alviola, P. A., Andrianangrasana, H., Brashares, J. S., Burton, A. C., Coronado, I., Corpuz, N., Enghoff, M., Fjeldsa, J., Funder, M., Holt, S., Hubertz, H., Jensen, A. E., Lewis, R., Massao, J., Mendoza, M. M., Ngaga, Y., Pipper, C. B., Poulson, M. K., Rueda, R. M., Sam, M. K., Skielbe, T., Sorensen, M., & Young, R. (2014): A multicountry assessment of tropical resource monitoring by local communities. *BioScience*, 64(3), 236-252.

David, C. G., Schulz, N., & Schlurmann, T. (2016). Assessing the application potential of selected ecosystem-based, low-regret coastal protection measures. In F. G. Renaud, K. Sudmeier-Rieux, M. Estrella, & U. Nehren (Eds.), *Ecosystem-based disaster risk reduction and adaptation in practice* (pp. 457-482), Cham, Switzerland: Springer.

Davidson, L. N. K., & Dulvy, N. K. (2017). Global marine protected areas to prevent extinctions. *Nature Ecology and Evolution* 1, 0040. https://doi.org/10.1038/s41559-016-0040

De Koning, F., Aguiñaga, M., Bravo, M., Chiu, M., Lascano, M., Lozada, T., & Suarez, L. (2011). Bridging the gap between forest conservation and poverty alleviation: the Ecuadorian Socio Bosque program. *Environmental Science and Policy*, 14(5), 531-542.

De Lisio, A. (2013): *La IIRSA o la integración física suramericana como dilema eco-sociopolítico. Doc. de trabajo*. Informe. *CLACSO*. Buenos Aires, Argentina. (pp. 236).

De Lisio, A. (2014): *Desarrollo sostenible/cambio climático/IIRSA: elementos de ecología política em América Latina y El Caribe. ENGOV Working Paper Series no. 9.* (pp. 23).

Decker, D. J., Raik, D., & Siemer, W. F. (2004). A Practitioner's Guide - Community-Based Deer Management. Cornell University, Ithaca, New York. *Northeast Wildlife Damage Management Research and Outreach Cooperative*. (pp. 56)

Depledge, D., & Dodds, K. (2017): Bazaar governance: situating the Arctic Circle. In K. Keil & S. Knecht (Eds.), *Governing Arctic Change* (pp. 141-160). London, UK: Palgrave MacMillan.

Di Marco, M., Butchart, S. H. M., Visconti, P., Buchanan, G. M., Ficetola, G. F., & Rondinini, C. (2015). Synergies and trade-offs in achieving global biodiversity targets. *Conservation Biology*, 30, 189-195.

Diegues, A. C. (2008). *Marine Protected Areas and Artisanal Fisheries in Brazil. Samudra Monograph. Ed. Anil Menon. International Collective in Support of Fisherworkers. India*. (pp. 68).

Doswald, N., & Estrella, M. (2015): Promoting Ecosystems for Disaster Risk Reduction and Climate Change Adaptation. *UNEP Discussion Paper*. Available at: https://wedocs.unep.org/handle/20.500.11822/14071.

Duarte, C. M., Losada, I. J., Hendriks, I. E., IMazarrasa, I., & Marbà, N. (2013). The role of coastal plant communities for climate change mitigation and adaptation. *Nature Climate Change*, 3, 961-968.

Duit, A., Galaz, V., Eckerberg, K., & Ebbesson, J. (2010). Introduction: Governance, complexity, and resilience. *Global Environmental Change*, 20, 363–368.

Durham W. H. (2008). Fishing for solutions: ecotourism and conservation in Galapagos National Park. In: A. Stronza & W. H. Durham (Eds.), *Ecotourism and Conservation in the Americas* (pp. 66-90). Wallingford, UK: Cabi International.

Durigan, G., Engel, V. L., Torezan, J. M., Melo, A. C. G., Marques, M. C. M., Martins, S. V., Reis, A., & Scarano, F. R. (2010). Normas jurídicas para a restauração ecológica: uma barreira a mais para dificultar o êxito das iniciativas? *Revista Árvore*, 34, 471–485.

Dyer, G. A., López-Feldman, A., Yúnez-Naude, A., & Taylor, J. E. (2014). Genetic erosion in maize's center of origin. *Proceedings of the National Academy of Sciences*, 111, 14094-14099.

Easterly, W. (2015). The SDGs should stand for senseless, dreamy, garbled. *Foreign Policy,* Available at: http://foreignpolicy.com/2015/09/28/the-sdgs-are-utopian-and-worthless-mdgs-development-rise-of-the-rest/

Echeverría, C., Smith-Ramírez, C., Aronson, J., & Barrera-Cataño, J. I. (2015). Good news from Latin America and the Caribbean: national and international restoration networks are moving ahead. *Restoration Ecology*, 23(1), 1–3.

ECLAC (2011): *Social Panorama of Latin America 2011. Briefing Paper, Economic Commission for Latin America and the Caribbean (ECLAC)*, Santiago, Chile, (pp. 49).

ECLAC (2013). *An assessment of the economic and social impacts of climate change on the agriculture sector in the Caribbean.* Economic Commission for Latin America and the Caribbean (S. Hutchinson, C. Gomes, D. Alleyne, & W. Phillips, Eds.). Port of Spain: United Nations.

ECLAC, (2016): *Economic Survey of Latin America and the Caribbean, 2016. Economic Commission for Latin America and the Caribbean*, Santiago, Chile.

Edgar, G. J., Stuart-Smith, R. D., Willis, T. J., Kininmonth, S., Baker, S. C., Banks, S., Barrett, N. S., Becerro, M. A., Bernard, A. T. F., Berkhout, J., Buxton, C. D., Campbell, S. J., Cooper, A. T., Davey, M., Edgar, S. C., Försterra, G., Galván, D. E., Irigoyen, A. J., Kushner, D. J., Moura, R., Parnell, P. E., Shears, N. T., Soler, G., Strain, E. M. A., & Thomson, R. J. (2014). Global conservation outcomes depend on marine protected areas with five key features. *Nature*, 506, 216-220.

Ellis, E. A., & Porter-Bolland, L. (2008). Is community-based forest management more effective than protected areas? A comparison of land use/land cover change in two neighboring study areas of the Central Yucatan Peninsula, Mexico. *Forest Ecology and Management*, 256, 1971-1983.

Emperaire, L. (2017) Saberes tradicionais e diversidade das plantas cultivadas na Amazônia. In B. Baptiste, D. Pacheco, M. Carneiro da Cunha, & S. Diaz (Eds.). *Knowing our Lands and Resources: Indigenous and Local Knowledge of Biodiversity and Ecosystem Services in the Americas.* Knowledges of Nature 11. (pp. 41-62). UNESCO: Paris.

Epanchin-Niell, R. S., & Wilen, J. E. (2014). Individual and cooperative management of invasive species in human-mediated landscapes. *American Journal of Agricultural Economics*, 97(1), 180–198.

Equihua Zamora, M., García Alaniz, N., Pérez-Maqueo, O., Benítez Badillo, G., Kolb, M., Schmidt, M., Equihua-Benítez, J., Maeda, P., & Álvarez Palacios, J. L. (2014). Integridad Ecológica como Indicador de la Calidad Ambiental. In C. A. González Zuarth, A. Vallarino, J. C. Pérez Jiménez, & A. M. Low Pfeng (Eds.), *Bioindicadores: Guardianes de nuestro futuro ambiental* (pp. 695–718). Mexico City: El Colegio de la Frontera Sur (ECOSUR).

Estrada-Carmona, N., Hart, A. K., DeClerck, F. A. J., Harvey, C. A., & Milder, J. C. (2014). Integrated landscape management for agriculture, rural livelihoods, and ecosystem conservation: an assessment of experience from Latin America and the Caribbean. *Landscape and Urban Planning*, 129, 1–11.

Estrella, M., Renaud, F. G., Sudmeier-Rieux, K., & Nehren, U. (2016). Defining new pathways for ecosystem-based disaster risk reduction and adaptation in the post-2015 Sustainable Development Agenda. In F. G. Renaud, K. Sudmeier-Rieux, M. Estrella, & U. Nehren (Eds.), *Ecosystem-Based Disaster Risk Reduction and Adaptation in Practice,* Advances in Natural and Technological Hazards Research 42 (pp. 553–591). Cham, Switzerland: Springer,

Evans, K., Murphy, L., & de Jong, W. (2014). Global versus local narratives of REDD: a case study from Peru's Amazon. *Environmental Science and Policy*, 35, 98-108.

Ezzine-de-Blas, D., Wunder, S., Ruiz-Pérez, M., & Moreno-Sanchez, R. d. P. (2016). Global patterns in the implementation of payments for environmental services. *PLoS One* 11(3), e0149847. https://doi.org/10.1371/ journal.pone.0149847

Fahlgren, N., Bart, R., Herrera-Estrella, L., Réllan-Álvarez, R., Chitwood, D. H., & Dinneny, J. R. (2016). Plant scientists: GM technology is safe. *Science 351*: 824.

Faith, D. P. (2014). Ecosystem services can promote conservation over conversion and protect local biodiversity, but these local win-wins can be a regional disaster. *Australian Zoologist*, 38, 477-487.

Faith, D. P. (2016). The PD Phylogenetic diversity framework: linking evolutionary history to feature diversity for biodiversity conservation. In R. Pellens & P. Grandcolas (Eds.), *Biodiversity Conservation and Phylogenetic Systematics* (pp. 39–56). Cham, Switzerland: Springer.

Fearnside, P. M. (1999). Biodiversity as an environmental service in Brazil's Amazonian forests: Risks, value and conservation. *Environmental Conservation*, 26(4), 305-321.

Fearnside, P. M. (2003). Conservation policy in Brazilian Amazonia: Understanding the dilemmas. *World Development*, 31(5), 757-779.

Fernandez-Gimenez, M. E., Ballard, H. L., & Sturtevant V. E. (2008). Adaptive management and social learning in collaborative and community-based monitoring: a study of five community-based forest organizations in the western USA. *Ecology and Society*, 13(2), 4.

Ferro, V. G., Lemes, P., Melo, A. S., & Loyola, R. (2014). The reduced effectiveness of protected areas under climate change threatens Atlantic Forest tiger moths. *PLoS One*, 9(9), e107792. https://doi.org/10.1371/journal.pone.0107792.

Fisher, J.A., Patenaude, G., Giri, K., Lewis, K., Meir, P., Pinho, P., Rounsevell, M. D. A., Williams, M. (2014). Understanding the relationships between ecosystem services and poverty alleviation: a conceptual framework. *Ecosystem Services*, 7, 34-45.

Fleming, A. H., & Pyenson, N. D. (2017). How to produce translational research to guide Arctic policy. *BioScience*, 67, 490-493.

Fogel, R. (2007). *Diversidad cultural y biodiversidad: El caso de las comunidades indígenas. In: Biodiversidad del Paraguay, uma aproximación a sus realidades.* D. A. Salas-Dueñas & J.F. Facetti (Eds.). Fundación Moisés Bertoni, USAID, GEF/BM..

Foley, J. A., Ramankutty, N., Brauman, K. A., Cassidy, E. S., Gerber, J. S., Johnston, M., Mueller, N. D., O'Connell, C., Ray, D. K., West, P. C., Balzer, C., Bennett, E. M., Carpenter, S. R., Hill, J., Monfreda, C., Polasky, S., Rockstrom, J., Sheehan, J., Siebert, S., Tilman, D., & Zaks, D. P. (2011). Solutions for a cultivated planet. *Nature*, 478, 337-342.

Folke, C., Hahn, T., Olsson, P., & Norberg, J. (2005). Adaptive governance of social-ecological systems. *Annual Reviews of Environmental Resources*, 30, 441–73.

Fonseca, C. R., Guadagnin, D. L., Emer, C., Masciadri, S., Germain, P., & Zalba, S. M. (2013). Invasive alien plants in the Pampas grasslands: a tri-national cooperation challenge. *Biological Invasions*, 15, 1751–1763.

Fonseca, P. (2015). A major center of biodiversity research crumbles. *Scientific American*, available at: https://www.scientificamerican.com/article/a-major-center-of-biodiversity-research-crumbles/

Franks, D. M., Davis, R., Bebbington, A. J., Ali, S. H., Kempa, D., & Scurrahg, M. (2014). Conflict translates environmental and social risk into business costs. *Proceedings of the National Academy of Sciences*, 111, 7576-7581.

Freitas, D. M., & Tagliani, R. A. (2009). The use of GIS for the integration of traditional and scientific knowledge in supporting artisanal fisheries management in southern Brazil. *Journal of Environmental Management*, 90, 2071-2080.

Fundación Natura. (2016). *WaterShared. Bolivia*. Available at: http://www.watershared.net/ ?page_id=12.

Fung, A., & Wright, E. O. (2001). Deepening democracy: Innovations in empowered participatory governance. *Politics and Society*, 29(1), 5-41.

Garcia, L. C., Ribeiro, D. B., Oliveira Roque, F. de, Ochoa-Quintero, J. M., & Laurance, W. F. (2017). Brazil's worst mining disaster: corporations must be compelled to pay the actual environmental costs. *Ecological Applications*, 27, 5–9.

Garcia, M. C. (2003): *Participación Publica: Herramientas de Participación para la Gestión Costera Patagónica*. Fundación Patagonia Natural. 59p.

García-Alaníz, N., Equihua, M., Pérez-Maqueo, O., Equihua Benítez, J., Maeda, P., Pardo Urrutia, F., Flores Martínez, J. J., Villela Gaytán, S. A., Schmidt, M. (2017). The Mexican national biodiversity and ecosystem degradation monitoring system. *Current Opinion in Environmental Sustainability*, 26, 62–68. https://doi.org/10.1016/j.cosust.2017.01.001

García-de-Lomas, J., & Vilà, M. (2015). Lists of harmful alien organisms: are the national regulations adapted to the global world? *Biological Invasions*, 17(11), 3081-3091.

Gardener, M. R, Bustamante, R. O., Herrera, I., Durigan, G., Pivello, V. R., Moro, M. F., Stoll, A., Langdon, B., Baruch, Z., Rico, A., Arredondo-Nuñez, A., & Flores, S. (2012). Plant invasions research in Latin America: fast track to a more focused agenda. *Plant Ecology and Diversity*, 5(2), 225-232.

Gardner, T. A., Von Hase, A., Brownlie, S., Ekstrom, J. M. M., Pilgrim, J. D., Savy, C. E., Stephens, R. T. T., Treweek, J., Ussher, G. T., Ward, G., & Kate, K. T. (2013). Biodiversity offsets and the challenge of achieving no net loss. *Conservation Biology*, 27, 1254-1264.

Garrett, R. D., Rueda, X., & Lambin, E. F. (2013). Globalization's unexpected impact on soybean production in South America: linkages between preferences for non-genetically modified crops, eco-certifications, and land use. *Environmental Research Letters*, 8(4), 44055. https://doi.org/10.1088/1748-9326/8/4/044055.

Gasparatos, A., Doll, C. N. H., Esteban, M., Ahmed, A. & Olang, T. A. (2017): Renewable energy and biodiversity: Implications for transitioning to a Green Economy. *Renewable and Sustainable Energy Reviews*, 70, 161–184.

GEF/UNEP/CABI, (2011): *Strategy and Action Plan for Invasive Alien Species in the Caribbean Region 2011-2016*. Trinidad and Tobago, 53 pp.

GEFIEO (2015). *Impact Evaluation of GEF Support to Protected Areas and Protected Area Systems: Highlights.* Global Environmental Facility Independent Evaluation Office, Washington, DC.

Gelcich, S., Vargas, C., Carreras, M. J., Castilla, J. C., Donlan, C. J. (2017). Achieving biodiversity benefits with offsets: Research gaps, challenges, and needs. *Ambio*, 46(2), 184–189.

Geldmann, J., Barnes, M., Coad, L., Craigie, I. D., Hockings, M., & Burgess, N. D. (2013). Effectiveness of terrestrial protected areas in reducing habitat loss and population declines. *Biological Conservation*, 161, 230–238.

Geneletti, D. (2016). Strengthening biodiversity and ecosystem services in impact assessment decisions for better decisions. In D. Geneletti (Ed.), *Handbook on Biodiversity and Ecosystem Services in Impact Assessment* (pp. 477-486) Cheltenham, UK: Edward Elgar Publishing.

Gerage, J. M., Meira, A. P. G. , & da Silva, M. V. (2017). Food and nutrition security: pesticide residues in food. *Nutrire*, 42, 3. https://doi.org/10.1186/s41110-016-0028-4.

Gerard, D. 2000. The law and economics of reclamation bonds. *Resources Policy*, 26, 189–197.

Geyer, L. L., & Lawler, D. (2016). Avoiding loss of agricultural subsidies: Swampbuster. In: C. M. Finlayson, M. Everard, K. Irvine, R. J. McInnes, B. A. Middleton, A. A. van Dam, N. C. Davidson (Eds.), *The Wetland Book* (pp. 1-4). Netherlands: Springer.

Gibbs, H. K., Rausch, L., Munger, J., Schelly, I., Morton, D. C., Noojipady, P., Soares-Filho, B., Barreto, P., Micol, L., & Walker, N. (2015). Brazil´s soy moratorium. *Science*, 347, 377-378.

Gibson, C. C, McKean, M. A., & Ostrom, E. (2000). *People and Forests: Communities, Institutions, and Governance*. Cambridge and London: The MIT Press (pp. 274).

Glowka, L. (2000). Bioprospecting, alien invasive species, and hydrothermal vents: three emerging legal issues in the conservation and sustainable use of biodiversity. *Tulane Environmental Law Journal*, 13(2), 329-360.

Goldman-Benner, R. L., Benitez, S., Boucher, T., Calvache, A., Daily, G., Kareiva, P., Kroeger, T., & Ramos, A. (2012). Water funds and payments for ecosystem services: practice learns from theory and theory can learn from practice. *Oryx*, 46(1), 55–63.

Goldstein, A., & Gonzalez, G. (2014). *Turning over a new leaf: state of the forest carbon markets 2014*. Forest Trends´ Ecosystem Marketplace. Available at: http://www.forest-trends.org/documents/files/doc_4770.pdf.

Gomar, J. O. V. (2014). International targets and environmental policy integration: The 2010 Biodiversity Target and its impact on international policy and national implementation in Latin America and the Caribbean. *Global Environmental Change*, 29, 202-214.

Gómez Lozano, R., Anderson, L., Akins, J. L., Buddo, D. S. A., Garcia Moliner, G., Gourdin, F., Laurent, M., Lilyestron, C., Morris Jr., J. A., Ramnanan, N., & Torres, R. (2013). *Estrategia regional para el control del Pez León Invasor en el Gran Caribe.* Iniciativa Internacional sobre los Arrecifes Coralinos. (pp. 32).

Gómez, I., & Méndez, V. E. (2005). *Associación de comunidades forestales de Petén, Guatemala: Contexto, logros y desafíos.* Prisma. (pp. 60).

Gonçalves, B., Marques, A., Soares, A. M. V. M., & Pereira, H. M. (2015). Biodiversity offsets: from current challenges to harmonized metrics. *Current Opinion in Environmental Sustainability*, 14, 61–67.

Gordillo Jordan, J. F., Hunt C., & Stronza, A. (2008). An ecotourism partnership in the Peruvian Amazon: the case of Posada Amazonas. In A. Stronza & W.H. Durham (Eds.), *Ecotourism and Conservation in the Americas* (pp. 30-48). Wallingford, UK: Cabi International.

Gray, N. J., Gruby, R. L., & Campbell, L. M. (2014): Boundary objects and global consensus: scalar narratives of marine conservation in the Convention on Biological Diversity. *Global Environmental Politics*, 14(3), 64-83.

Grima, N., Singh, S. J., Smetschka, B., & Ringhofer, L. (2016): Payment for ecosystem services (PES) in Latin America: analyzing the performance of 40 case studies. *Ecosystem Services*, 17, 24–32.

GSIA (2016). *2016 Global Sustainable Investment Review.* Global Sustainable Investment Alliance. Available at: http://www.gsi-alliance.org/wp-content/uploads/2017/03/ GSIR_Review 2016.F.pdf
Guariguata, M. R., Sist, P., & Nasi, R. (2012). Multiple use management of tropical production forests: how can we move from concept to reality? *Forest Ecology and Management*, 268, 1-5.
Guedes, G.R., L.K. VanWey, J.R. Hull, M. Antigo, and A.F. Barbieri. (2014). Poverty dynamics, ecological endowments, and land use among smallholders in the Brazilian Amazon. *Social Science Research*, 43, 74-91.
Gunter, U., Ceddia, M. G., & Tröster, B. (2017). International ecotourism and economic development in Central America and the Caribbean. *Journal of Sustainable Tourism*, 25, 43-60. https://doi.org/10.1080/09669582.2016.1173043
Hajjar, R., Oldekop, J. A., Cronkleton, P., Etue, E., Newton, P., Russel, A. J. M., Tjajadi, J. S., Zhou, W., & Agrawal, A. (2017). The data not collected on community forestry. *Conservation Biology*, 30, 1357-1362.
Hamrick, K. (2015) *Ahead of the curve. State of the Voluntary Carbon Markets 2015.* Forest Trends´s Ecosystem Marketplace. Available at: https://www.forest-trends.org/publications/ahead-of-the-curve/
Handford, C. E., Elliott, C. T., & Campbell, K. (2015). A review of the global pesticide legislation and the scale of challenge in reaching the global harmonization of food safety standards. *Integrated Environmental Assessment and Management*, 9999, 1–12.
Hanson, C., Talberth, J., & Yonavjak, L. (2011). Forests for Water: Exploring Payments for Watershed Services in the U.S. South. Southern Forests for the Future Incentives Series (WRI Issue Brief 2), Washington, DC.: World Resources Institute. Available at http://www.wri.org/publication/ forests-water.
Hardin, G. (1968). The tragedy of the commons. *Science*, 162, 1243-1248.
Harris, J. B. C., Reid, J. L., Scheffers, B. R., Wanger, T. C., Sodhi, N. S., Fordham, D. A., & Brook, B. W. (2012). Conserving imperiled species: a comparison of the IUCN Red List and U.S. Endangered Species Act. *Conservation Letters*, 5(1), 64-72.
Haslam, P. A., & Tanimoune, N. A. (2016). The determinants of social conflict in the Latin American mining sector: new evidence with quantitative data. *World Development*, 78, 401-419.
Hidasi-Neto, J., Loyola, R. D., & Cianciaruso, M. V. (2013). Conservation actions based on red lists do not capture the functional and phylogenetic diversity of birds in Brazil. *PLoS One*, 8, e73431. https://doi.org/10.1371/journal.pone.0073431
Hilbeck, A., Binimelis, R., Defarge, N., Steinbrecher, R., Székács, A., Wickson, F., Antoniou, M., Bereano, P. L, Clark, E. A., Hansen, M., Novotny, E., Heinemann, J., Meyer, H., Shiva, V., & Wynne, B. (2015). No scientific consensus on GMO safety. *Environmental Sciences Europe*, 27, 4.
Hilty, J. A., Lidicker Jr, W. Z., & Merenlender, A. (2006). *Corridor Ecology*. Washington, US: Island Press (pp. 344).
Hoffmann, D., Oetting, I., Arnillas, C. A., & Ulloa, R. (2011). Climate change and protected areas in the Tropical Andes. In S. K. Herzog, R. Martínez, P. M. Jørgensen, & H. Tiessen (Eds.), *Climate Change and Biodiversity in the Tropical Andes* (pp. 311-325). São José dos Campos, Brazil: Inter-American Institute for Global Change Research (IAI); Paris: Scientific Committee on Problems of the Environment (SCOPE).
Hogdon, B. D., Hghell, D., Ramos, V. H., & McNab, R. B. (2015). *Deforestation trends in the Maya Biosphere Reserve*. Guatemala: USAID. 14p.
Holden, M., (2013). Sustainability indicator systems within urban governance: Usability analysis of sustainability indicator systems as boundary objects. *Ecological Indicators*, 32, 89-96.
Hrabanski, M., (2015). The biodiversity offsets as market-based instruments in global governance: Origins, success and controversies. *Ecosystem Services*, 15, 143-151.
Hsu, A., Alexandre, N, Cohen, S., Jao, P, Khusainova, E, Mosteller, D, Peng, Y, Rosengarten, C, Schwartz, J. D., Spawn, A., Weinfurter, A., Xu, K., Yin, D., Ivanenko, M., Cook, A. R., Foo, J. M., Yi, J., Sarathy, J., Torres-Quintanilla, D., Wong, D., Yick, C., Young, A., Zomer, A., Hirsch, P., Householder, A., Nestor, C., Shah, S., Song, Y., Yan, C., Levy, M. A., de Sherbinin, A., Jaiteh, M., Chai-Onn, T., Esty, D. C.,

Dale, L., Rosengarten, C., Rosengarten, T., Seven, H., & Weeks, C. (2016): *2016 Environmental Performance Index*. New Haven, CT: Yale University. Available: www.epi.yale.edu.

Hubeau, M., Marchand, F., Coteur, I., Mondelaers, K., Debruyne, L., & Van Huylenbroeck, G. (2017): A new agri-food systems sustainability approach to identify shared transformation pathways towards sustainability. *Ecological Economics*, 131, 52-63.

INE (2011). *Compendio Estadístico Ambiental de Guatemala (2010).* Instituto Nacional de Estadística (INE), Sección de Estadísticas Ambientales, Oficina Coordinadora Sectorial de Estadísticas de Ambiente y Recursos Naturales, OCSE/Ambiente, Ciudad de Guatemala, Guatemala (pp. 357).

IPBES (2016): *Information on work related to policy support tools and methodologies (deliverable 4 (c)) (IPBES/4/INF/14). Kuala Lumpur: Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services*. Retrieved from http://www.minambiente.it/sites/default/files/archivio/allegati/biodiversita/ ipbes_3_5_advance.pdf

IPCC (2014). Summary for policymakers. In C. B. Field, V. R. Barros, D. J. Dokken, K. J. Mach, M. D. Mastrandrea, T. E. Bilir, M. Chatterjee, K. L. Ebi, Y. O. Estrada, R. C. Genova, B. Girma, E. S. Kissel, A. N. Levy, S. MacCracken, P. R. Mastrandrea, & L. L. White (Eds.), *Climate Change 2014: Impacts, Adaptation, and Vulnerability. Part A: Global and Sectoral Aspects. Contribution of Working Group II to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change* (pp. 1-32). Cambridge: Cambridge University Press.

IPEA (2012). *Quilombos das Américas: articulação de comunidades afrorrurais : documento síntese*. Brasília: Instituto de Pesquisas Econômicas Aplicadas (pp. 79).

IUCN (2007). *Connectivity conservation: international experience in planning, establishment and management of biodiversity corridors*. Switzerland: WCPA/ International Union for the Conservation of Nature (pp. 19)

IUCN (2016). *A global standard for the identification of Key Biodiversity Areas*. International Union for the Conservation of Nature. Available at https://portals.iucn.org/library/node/46259.

Jacobs, K. R., Nicholson, L., Murry, B. A., Maldonado-Román, M., & Gould, W. A. (2016). Boundary organizations as an approach to overcoming science-delivery barriers in landscape conservation: a Caribbean case study. *Caribbean Naturalist*, Special Issue No. 1, 87–107.

Jacobsen, S.-E., Sørensen, M., Pedersen, S. M., & Weiner, J. (2013). Feeding the world: genetically modified crops versus agricultural biodiversity. *Agronomy for Sustainable Development*, 33, 651. https://doi.org/10.1007/s13593-013-0138-9.

Jantz, P., Goetz, S. J., & Laporte, N. (2014). Carbon stock corridors to mitigate climate change and promote biodiversity in the tropics. *Nature Climate Change*, 4, 138–142.

Jaskoski, M. (2014). Environmental licensing and conflict in Peru's mining sector: a path-dependent analysis. *World Development*, 64, 873–883.

Jenkins, C.N., & Joppa, L. (2009). Expansion of the global terrestrial protected area system. *Biological Conservation*, 142, 2166–2174.

Jones, N. F., Pejchar, L., & Kiesecker, J. M. (2015). The energy footprint: how oil, natural gas, and wind energy affect land for biodiversity and the flow of ecosystem services. *Bioscience*, 65, 290-301.

Jørgensen, D. (2015). Ecological restoration as objective, target, and tool in international biodiversity policy. *Ecology and Society*, 20(4). https://doi.org/10.5751/ES-08149-200443.

Jørgensen, D., Nilsson, C., Hof, A. R., Hasselquist, E. M., Baker, S., Chapin III, F. S., Eckerberg, K., Hjältén, J., Polvi, L., & Meyerson, L. A. (2014). Policy language in restoration ecology. *Restoration Ecology*, 22(1), 1-4.

Joshi, D. K., Hughes, B. B., & Sisk, T. D. (2015). Improving governance for the post-2015 Sustainable Development Goals: scenario forecasting the next 50 years. *World Development*, 70, 286-302.

Juhola, S., Glaas, E., Linnér, B.-O., & Neset, T.-S. (2016). Redefining maladaptation. *Environmental Science and Policy*, 55, 135–140.

Juma, C., & Gordon, K. (2014). Transgenic crops and food security. In A. Ricroch & S. J. Fleischer (Eds.). *Plant Biotechnology: Experience and Future Prospects* (pp. 45-58). Switzerland: Springer.

Kalikoski, D. C., Seixas, C. S., & Almudi, T. (2009). Gestão compartilhada e comunitária da pesca no Brasil: Avanços e desafios. *Ambiente e Sociedade*, 12(1), 151-172.

Kareiva, P., & Levin, S. A. (Eds.) (2003). *The Importance of Species: Perspectives on Expendability and Triage*. Princeton, US, and Oxford, UK: Princeton University Press.

Kasecker, T. P., Ramos-Neto, M. B., Silva, J. M. C. da, & Scarano, F. R. (2017). Ecosystem-based adaptation to climate change: defining hotspot municipalities for policy design and implementation in Brazil. *Mitigation and Adaptation Strategies to Global Change* https://doi.org/10.1007/s11027-017-9768-6.

Kauffman, C. M., & Martin, P. L. (2014). Scaling up Buen Vivir: globalizing local environmental governance from Ecuador. *Global Environmental Politics*, 14(1), 40-58.

Kaufmann, D. (2015). Corruption matters. *Finance & Development*, 52(3), 20-23.

Kearney, J., Berkes, F., Charles, A., Pinkerton, E., & Wiber, M. (2007). The role of participatory governance and community-based management in integrated coastal and ocean management in Canada. *Coastal Management*, 35, 79-104.

Keith, D. A., Rodríguez, J. P., Brooks, T. M., Burgman, M. A., Barrow, E. G., Bland, L., Comer, P. J., Franklin, J., Link, J., McCarthy, M. A., Miller, R. M., Murray, N. J., Nel, J., Nicholson, E., Oliveira-Miranda, M. A., Regan, T. J., Rodríguez-Clark, K. M., Rouget, M., & Spalding, M. D. (2015). The IUCN Red List of Ecosystems: motivations, challenges, and applications. *Conservation Letters*, 8(3), 214–226.

Kellert, S. R., Mehta, J. N., Ebbin, S. A., & Lichtenfeld, L. L. (2000). Community natural resources management: Promise, rhetoric, and reality. *Society and Natural Resources*, 13, 705-715.

Kennicutt, M. C. II, Chown, S. L., Cassano, J. J., Liggett, D., Massom, R., Peck, L. S., Rintoul, S. R., Storey, J. W. V., Vaughan, D. G., Wilson, T. J., & Sutherland, W. J. (2014). Six priorities for Antarctic science. *Nature*, 512, 23-25.

Khoury, C. K., Achicanoy, H. A., Bjorkman, A. D., Navarro-Racines, C., Guarino, L., Flores-Palacios, X., Engels, J. M. M., Wiersema, J. H., Dempewolf, H., Sotelo, S., Ramírez-Villegas, J., Castañeda-Álvarez, N. P., Fowler, C., Jarvis, A., Rieseberg, L. H., & Struik, P. C. (2016). Origins of food crops connect countries worldwide. *Proceedings of the Royal Society B*, 283, 20160792.

Klak, T., (2007). Sustainable ecotourism development in Central America and the Caribbean: review of debates and conceptual reformulation*.* *Geography Compass*, 1/5, 1037–1057.

Klimas, C. A., Kainer, K. A., & Wadt, L. H. d. O. (2012). The economic value of sustainable seed and timber harvests of multi-use species: an example using *Carapa guianensis*. *Forest Ecology and Management*, 268, 81-91.

Knowles, J. E., Doyle, E., Schill, S. R., Roth, L. M., Milam, A., & Raber, G. T. (2016). Establishing a marine conservation baseline for the insular Caribbean. *Marine Policy*, 60, 84-97.

Kostyack, J., Lawler, J. J., Goble, D. D., Olden, J. D., & Scott, J. M. (2011). Beyond reserves and corridors: policy solutions to facilitate the movement of plants and animals in a changing climate. *Bioscience*, 61, 713-719.

Kothari, A., DeMaria, F., & Acosta, A. (2014). Buen vivir, degrowth and ecological swaraj: Alternatives to sustainable development and the green economy. *Development*, 57(3–4), 362–375.

Koury, C. G., & Guimarães, E. R. (2012). *O Desafio da Gestão Participativa, Oportunidades: A Experiência na RDS Uatumã.* Série Integração, Transformação e Desenvolvimento: Áreas Protegidas e Biodiversidade. Rio de Janeiro: Fundo Vale para o Desenvolvimento Sustentável.

KPMG (2013): *The KPMG Survey of Corporate Responsibility Reporting 2013.* KPMG International Cooperative.

KPMG (2015). T*he KPMG Survey of Corporate Responsibility Reporting 2015. International Cooperative.* The Netherlands: Haymarket Network Ltd.. Available at: https://assets.kpmg.com/content/dam/kpmg/pdf/2016/02/kpmg-international-survey-of-corporate-responsibility-reporting-2015.pdf

Krüger, O. (2005). The role of ecotourism in conservation: panacea or Pandora's box? *Biodiversity and Conservation*, 14, 579–600.

Kumar, P., Brondizio, E., Gatzweiler, F., Gowdy, J., de Groot, D., Pascual, U., Reyers, B., & Sukhdev, P. (2013): The economics of ecosystem services: from local analysis to national policies. *Current Opinion in Environmental Sustainability*, 5, 78-86.

Lago Candeira, A., and Silvestri, L. (2013). *Análisis del Marco Legal de Panamá para la Implementación del Protocolo de Nagoya*. Informe Proyecto Regional PNUMA/GEF ABS LAC. Madrid, España: Cátedra UNESCO de Territorio y Medio Ambiente (pp. 51).

Lambin, E. F., Meyfroidt, P., Rueda, X., Blackman, A., Börner, J., Cerutti, P. O., Dietsch, T., Jungmann, L., Lamarque, P., Lister, J., Walker, N. F., & Wunder, S. (2014). Effectiveness and synergies of policy instruments for land use governance in tropical regions. *Global Environmental Change*, 28, 129–140.

Lange, W., Pirzer, C., Dünow, L., & Schelchen, A. (2016). Risk perception for participatory ecosystem-based adaptation to climate change in the Mata Atlântica of Rio de Janeiro State, Brazil. In F. G. Renaud, K. Sudmeier-Rieux, M. Estrella, & U. Nehren (Eds.), *Ecosystem-based disaster risk reduction and adaptation in practice* (pp. 483-506). Switzerland: Springer.

Langholz, J. (1996). Economics, objectives, and success of private nature reserves in Sub-Saharan Africa and Latin America. *Conservation Biology*, 10(1), 271-280.

Larson, A. M. (2003). Decentralisation and forest management in Latin America: towards a working model. *Public Administration and Development*, 23, 211-226.

Larson, A. M. (2005). Democratic decentralisation in the forestry sector: Lessons learned from Africa, Asia and Latin America. In C. J. P. Colfer & D. Capistrano (Eds.), *The Politics of Decentralisation: Forests, People and Power* (pp. 32-62). New York: CIFOR, Earthscan.

Lautenbach, S., Seppelt, R., Liebscher, J., & Dormann, C. F. (2012). Spatial and temporal trends of global pollination benefit. *PLoS One*, *7*(4): e35954. https://doi.org/10.1371/journal.pone.0035954.

Le Gouvello, R., Hochart, L.-E., Laffoley, D., Simard, F., Andrade, C., Angel, D., Callier, M., De Monbrison, D., Fezzardi, D., Haroun, R., Harris, A., Hughes, A., Massa, F., Roque, E., Soto, D., Stead, S., & Marino, G. (2017). Aquaculture and marine protected areas: Potential opportunities and synergies. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 27 (S1), 1–13.

Le Velly, G., Dutilly, C., Ezzine de Blas, D., & Fernandez, C., (2015). *PES as Compensation? Redistribution of Payments for Forest Conservation in Mexican Common Forests*. Études et Documents, n° 28, *CERDI*. http://cerdi.org/production/show/id/1758/type_ production_id/1

Leadley, P., Proença, V., Fernández-Manjarrés, J., Pereira, H. M., Alkemade, R., Biggs, R., Bruley, E., Cheung, W., Cooper, D., Figueiredo, J., Gilman, E., Guénette, S., Hurtt, G., Mbow, C., Oberdorff, T., Revenga, C., Scharlemann, J. P. W., Scholes, R., Smith, M. S., Sumaila, U. R., & Walpole, M. (2014). Interacting regional-scale regime shifts for biodiversity and ecosystem services. *Bioscience*, 64, 665–679.

Lemes, P., Melo, A. S., & Loyola, R. D. (2013). Climate change threatens protected areas of the Atlantic Forest. *Biodiversity Conservation*, 23, 357–368.

Lemos, M. C., & Agrawal, A. (2006). Environmental Governance. *Annual Review of Environmental Resources*, 31, 297–325.

Lemos, M. C., Lob, Y. -J., Nelson, D. R., Eakin, H., & Bedran-Martins, A. M. (2016). Linking development to climate adaptation: Leveraging generic and specific capacities to reduce vulnerability to drought in NE Brazil. *Global Environmental Change*, 39, 170-179.

Lenschow, A., Newig, J., & Challies, E. (2015). Globalization's limits to the environmental state? Integrating telecoupling into global environmental governance. *Environmental Politics*, 25(1), 136-159. http://dx.doi.org/10.1080/09644016.2015.1074384

Lenzen, M., Moran, D., Kanemoto, K., Foran, B., Lobefaro, L., & Geschke, A. (2012). International trade drives biodiversity threats in developing nations. *Nature*, 486, 109-112.

Levis, C., Costa, F. R. C., Bongers, F., Peña-Claros, M., Clement, C. R., Junqueira, A. B., Neves, E. G., Tamanaha, E. K., Figueiredo, F. O. G., Salomão, R. P., Castilho, C. V., Magnusson, W. E., Phillips, O. L., Guevara, J. E., Sabatier, D., Molino, J.-F., Cárdenas López, D., Mendoza, A. M., Pitman, N. C. A., Duque, A., Núñez Vargas, P., Zartman, C. E., Vasquez, R., Andrade, A., Camargo, J. L., Feldpausch, T. R., Laurance, S. G. W., Laurance, W. F., Killeen, T. J., Nascimento, H. E. M, Montero, J. C., Mostacedo, B., Amaral, I. L., Vieira, I. C. G., Brienen, R., Castellanos, H., Terborgh, J., Carim, M. J. V., Guimarães, J. R. S., Coelho, L. S., Matos, F. D. de A., Wittmann, F., Mogollón, H. F., Damasco, G., Dávila, N., García-Villacorta, R., Coronado, E. N. H., Emilio, T., Lima Filho, D. de A., Schietti, J., Souza, P., Targhetta, N., Comiskey, J. A., Marimon, B. S., Marimon

Jr., B.-H., Neill, D., Alonso, A., Arroyo, L., Carvalho, F. A., Souza, F. C. de, Dallmeier, F., Pansonato, M. P., Duivenvoorden, J. F., Fine, P. V. A., Stevenson, P. R., Araujo-Murakami, A., Aymard C., G. A., Baraloto, C., do Amaral, D. D., Engel, J., Henkel, T. W., Maas, P., Petronelli, P., Cardenas Revilla, J. D., Stropp, J., Daly, D., Gribel, R., Ríos Paredes, M., Silveira, M., Thomas-Caesar, R., Baker, T. R., da Silva, N. F., Ferreira, L. V., Peres, C. A., Silman, M. R., Cerón, C., Valverde, F. C., Di Fiore, A., Jimenez, E. M., Peñuela Mora, M. C., Toledo, M., Barbosa, E. M., Bonates, L. C. de M., Arboleda, N. C., Farias, E. de S., Fuentes, A., Guillaumet, J. -L., Møller Jørgensen, P., Malhi, Y., Andrade Miranda, I. P. de, Phillips, J. F., Prieto, A., Rudas, A., Ruschel, A. R., Silva, N., von Hildebrand, P., Vos, V. A., Zent, E. L., Zent, S., Cintra, B. B. L., Nascimento, M. T., Oliveira, A. A., Ramirez-Angulo, H., Ramos, J. F., Rivas, G., Schöngart, J., Sierra, R., Tirado, M., van der Heijden, G., Torre, E. V., Wang, O., Young, K. R., Baider, C., Cano, A., Farfan-Rios, W., Ferreira, C., Hoffman, B., Mendoza, C., Mesones, I., Torres-Lezama, A., Medina, M. N. U., van Andel, T. R., Villarroel, D., Zagt, R., Alexiades, M. N., Balslev, H., Garcia-Cabrera, K., Gonzales, T.,Hernandez, L., Huamantupa-Chuquimaco, I., Manzatto, A. G., Milliken, W., Cuenca, W. P., Pansini, S., Pauletto, D., Arevalo, F. R., Costa Reis, N. F., Sampaio, A. F., Urrego Giraldo, L. E., Valderrama Sandoval, E. H., Valenzuela Gamarra, L., Vela, C. I. A., ter Steege, H. (2017). Persistent effects of pre-Columbian plant domestication on Amazonian forest composition. *Science*, 355 (6328), 925-931.

Levrel, H., Pioch, S., & Spieler, R. (2012). Compensatory mitigation in marine ecosystems: Which indicators for assessing the “no net loss” goal of ecosystem services and ecological functions? *Marine Policy*, 36 (6), 1202–1210.

Lewis, E., Pinchot, A., & Christianson, G. (2016). *Navigating the Sustainable Investment Landscape.* Washington, DC: World Resources Institute (WRI). Retrieved from https://www.wri.org/sites/default/files/Navigating_the_Sustainable_Investment_Landscape.pdf

Lewis-Lettington, R. J., Müller, M. R., Young, T. R., Nnadozie, K. A., Halewood, M. & Cabrera Medaglia, J. (2006). *Methodology for Developing Policies and Laws for Access to Genetic Resources and Benefit Sharing*. Rome, Italy: International Plant Genetic Resources Institute. (pp. 35).

Lima, D., & Pozzobon, J. (2005). Amazônia Sociambiental: sutentabilidade ecológica e diversidade social. *Estudos Avançados*, 19(54), 45-76.

Lima, M. G. B., Visseren-Hamakers, I. J., Braña-Varela, J., Gupta, A. (2017). A reality check on the landscape approach to REDD+: Lessons from Latin America. *Forest Policy and Economics*, 78, 10-20.

Liu, J., Calmon, M., Clewell, A., Liu, J., Denjean, B., Engel, V. L., & Aronson, J. (2017). South–south cooperation for large-scale ecological restoration. *Restoration Ecology*, 25, 27-32.

Liu, J., Hull, V., Batistella, M., DeFries, R., Dietz, T., Fu, F., Hertel, T. W., Izaurralde, R. C., Lambin, E. F., Li, S., Martinelli, L. A., McConnell, W. J., Moran, E. F., Naylor, R., Ouyang, Z., Polenske, K. R., Reenberg, A., Rocha, G. de M., Simmons, C. S., Verburg, P. H., Vitousek, P. M., Zhang, F., & Zhu, C. (2013). Framing sustainability in a telecoupled world. *Ecology and Society*, 18(2), 26. https://doi.org/10.5751/ES-05873-180226.

Liverman, D. M., & Vilas, S. (2006). Neoliberalism and the environment in Latin America. *Annual Review Environment and Resources*, 31, 327–363.

Locatelli, B., Evans, V., Wardell, A., Andrade, A., & Vignola, R. (2011). Forest and climate change in Latin America: linking adaptation and mitigation. *Forests*, *2*, 431-450.

Lomborg, B. (2017). *The mismeasure of development. Project Syndicate*, Available at: https://www.project-syndicate.org/commentary/sustainable-development-scorecards-problems-by-bjorn-lomborg-2017-10.

Lopes, P. F. M., Rosa, E. M., Salyvonchyk, S., Nora, V., & Begossi, A. (2013). Suggestions for fixing top-down coastal fisheries management through participatory approaches. *Marine Policy*, 40, 100-110.

Lowitt, K., Hickey, G. M., Saint Ville, A., Raeburn, K., Thompson-Colon, T., Laszlo, S., & Phillip, L. E. (2015). Factors affecting the innovation potential of smallholder farmers in the Caribbean Community. *Regional Environmental Change,* 15(7), 1367–1377.

Loyola, R. D., Lemes, P., Brum, F. T., Provete, D. B., & Duarte, L. D. S. (2014). Clade-specific consequences of climate change to amphibians in Atlantic Forest protected areas. *Ecography*, 37, 65–72.

Loyola, R. D., Nabout, J. C., Trindade-Filho, J., Lemes, P., Urbina-Cardona, J. N., Dobrovolski, R., Sagnori, M. D., & Diniz-Filho, J. A. F. (2012). Climate change might drive species into reserves: a case study of the American bullfrog in the Atlantic Forest Biodiversity Hotspot. *Alytes 29*, 61–74.

Lubell, M., Jasny, L., & Hastings, A. (2017). Network governance for invasive species management. *Conservation Letters*, 10(6), 699-707.

Lucas, P. L., Kok, M. T. J., Nilsson, M., Alkemade, R. (2014). Integrating biodiversity and ecosystem services in the post-2015 development agenda: goals structure, target areas and means of implementation. *Sustainability*, 6, 193-216.

Lui, G. V., & Coomes, D. A. (2015). Tropical nature reserves are losing their buffer zones, but leakage is not to blame. *Environmental Research*, 147, 580-589.

Luzar, J. B., Silvius, K. M., Overman, H., Giery, S. T., Read, J. M., & Fragoso, J. M. V. (2011). Large-scale environmental monitoring by indigenous peoples. *BioScience*, 61(10), 771-781.

Maccord, P. F. L., Silvano, R. A. M., Ramires, M. S., & Begossi, M. C. (2007). Dynamics of artisanal fisheries in two Brazilian Amazonian reserves: implications to co-management. *Hydrobiologia*, 583, 365-376.

Mace, G. M., Collar, N. J., Gaston, K. J., Hilton-Taylor, C., Akçakaya, H. R., Leader-Williams, N., Milner-Gulland, E. J., & Stuart, S. N. (2008). Quantification of extinction risk: IUCN's system for classifying threatened species. *Conservation Biology*, 22, 1424-1442.

Mach, M. E., Wedding, L. M., Reiter, S. M., Micheli, F., Fujita, R. M., & Martone, R. G. (2017). Assessment and management of cumulative impacts in California's network of marine protected areas. *Ocean and Coastal Management*, 137, 1-11.

Magrin, G. O., Marengo, J. A., Boulanger, J. -P., Buckeridge, M. S., Castellanos, E., Poveda, G., Scarano, F. R., & Vicuña, S. (2014). Central and South America. In V. R. Barros, C. B. Field, D. J. Dokken, M. D. Mastrandrea, K. J. Mach, T. E. Bilir, M. Chatterjee, K. L. Ebi, Y. O. Estrada, R. C. Genova, B. Girma, E. S. Kissel, A. N. Levy, S. MacCracken, P. R. Mastrandrea, & L. L. White (Eds.). *Climate Change 2014: Impacts, Adaptation, and Vulnerability. Part B: Regional Aspects. Contribution of Working Group II to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change* (pp. 1499-1566). Cambridge, United Kingdom and New York, NY, USA: Cambridge University Press.

Mamani Machaca, E. F. (2017). Saberes y conocimientos del pueblo indígena del Ayllu Sullka del municipio de Tomave, Potosí, Bolivia. In B. Baptiste, D. Pacheco, M. Carneiro da Cunha, & S. Diaz (Eds.). *Knowing our Lands and Resources: Indigenous and Local Knowledge of Biodiversity and Ecosystem Services in the Americas.* Knowledges of Nature 11 (pp. 96-104). UNESCO: Paris.

Marinaro, M. S., Grau, H. R., Macchi, L. & Zelaya, P. V. (2015). Land tenure and biological communities in dry Chaco forests of northern Argentina. *Journal of Arid Environments*, 123, 60-67.

Maron, M., Hobbs, R. J., Moilanen, A., Matthews, J. W., Christie, K., Gardner, T. A., Keith, D. A., Lindenmayer, D. B., & McAlpine, C. A. (2012). Faustian bargains? Restoration realities in the context of biodiversity offset policies. *Biological Conservation*, 155, 141–148.

Martin, A., McGuire, S., & Sullivan, S. (2013). Global environmental justice and biodiversity conservation. *The Geographical Journal*, 179(2), 122–131.

Martin, T. G., & Watson, J. E. M. (2016). Intact ecosystems provide best defence against climate change. *Nature Climate Change*, 6, 122-124.

Martin-Ortega, J., Ojea, E., & Roux, C. (2013). Payments for water ecosystem services in Latin America: a literature review and conceptual model. *Ecosystem Services*, 6, 122-132.

Martins, A. R. P., Ferraz, F. T., & da Costa, M. M. (2006). Sustentabilidade ambiental como nova dimensão do Índice de Desenvolvimento Humano dos países. *Revista do BNDES*, 13(26), 139-162.

Mas, A. H., & Dietsch, T. V. (2004). Linking shade coffee certification to biodiversity conservation: butterflies and birds in Chiapas, Mexico. *Ecological Applications*, 14(3), 642–654.

Matheus, F. S, & Raimundo, S. (2016). Public use and ecotourism policies in Brazilian and Canadian protected areas. *Études Caribéennes*, 33- 34 (online). https://doi.org/10.4000/etudescaribeennes.9344

Matthews, J. W., & Endress, A.G. (2008). Performance criteria, compliance success, and vegetation development in compensatory mitigation wetlands. *Environmental Management*, 41, 130–141.

May, P., Fernandes, L. S., & Rodríguez Osuna, V. (in press). Evolution of public policies and local innovation in landscape conservation in Rio de Janeiro. In U. Nehren, S. Schlueter, C. Raedig, D. Sattler, & H. Hissa (Eds.), *Strategies and tools for a sustainable Rio de Janeiro* (in press). Cham, Switzerland: Springer.

McGrath, D. G., Castro, F., Futemma, C., Amaral, B. D., & Calabria, J. (1993). Fisheries and the evolution of resource management on the Lower Amazon Floodplain. *Human Ecology*, 21(2), 167-198.

McGray, H., Hammill, A., & Bradley, R. (2007). *Weathering the Storm - Options for Framing Adaptation and Development.* Washington, DC, US: World Resources Institute.

McSweeney, K., Nielsen, E. A., Taylor, M. J., Wrathall, D. J., Pearson, Z., Wang, O., & Plumb, S. T. (2014). Drug policy as conservation policy: narco-deforestation. *Science*, 343, 489-490.

Medeiros, R., Young, C. E. F., Pavese, H. B., & Araújo, F. F. S. (2011). *Contribuição das Unidades de Conservação Brasileiras para a Economia Nacional: Sumário Executivo*. Brasília: UNEP-WCMC (pp. 44).

Meffe, G. K., Nielsen, L. A., Knight, R. L., & Schenborn, D. A. (2002). *Ecosystem Management - Adaptive Community-Based Conservation*. Washington: Island Press (pp. 333).

Mercer, J., Kelman, I., Alfthan, B., & Kurvits, T. (2012). Ecosystem-based adaptation to climate change in Caribbean small island developing states: integrating local and external knowledge. *Sustainability*, 4, 1908-1932.

Millar, C., & Fagre, D. (2007). Monitoring alpine plants for climate change: the North American GLORIA Project. *Cirmount, Mountain Views,* 1/1, 12-15.

Minambiente (2016). *Informe de Gestión*. Ministerio de Ambiente y Desarollo Sostenible. Available at http://www.minambiente.gov.co/images/planeacion-y-seguimiento/pdf/Informes_de_Gesti%C3%B3n/Informe_de_Gesti%C3%B3n_MADS/Informe_de_Gesti%C3%B3n_MADS_2016_1.pdf

Miteva, D. A., Pattanayak, S. K., & Ferraro, P. J. (2012). Evaluation of biodiversity policy instruments: what works and what doesn't? *Oxford Review of Economic Policy*, 28(1), 69-92.

Mittermeier, R. A., Gil, P. R., Hoffmann, M., Pilgrim, J., Brooks, T., Mittermeier, C. G., Lamoreux, J., & Fonseca, G. A. B. (2005). *Hotspots Revisited: Earth's Biologically Richest and Most Endangered Terrestrial Ecoregions*. 2nd ed.. Arlington, VA, USA: Conservation International (pp. 392).

Mittermeier, R. A., Mittermeier, C. G., Robles Gil, P., Pilgrim, J., Fonseca, G. A. B., Konstant, W. R., & Brooks, T. (2002). *Wilderness: Earth's Last Wild Places*. Arlington, VA, USA: Conservation International (pp. 576).

Montagnini, F., & Finney, C. (2011). Payments for environmental services in Latin America as a tool for restoration and rural development. *Ambio*, 40(3), 285-297.

Montoya, J. M., Donohue, I., & Pimm, S. L. (2018). Planetary boundaries for biodiversity: Implausible science, pernicious policies. *Trends in Ecology and Evolution*, 33(2), 71-73. https://doi.org/10.1016/j.tree.2017.10.004

Moran, D., & Kanemoto, K. (2017). Identifying species threat hotspots from global supply chains. *Nature Ecology and Evolution*, 1, 0023, https://doi.org/10.1038/s41559-016-0023.

Morton, D. C., Noojipady, P., Macedo, M. M., Gibbs, H., Victoria, D. C., Bolfe, E. L. (2016). Reevaluating suitability estimates based on dynamics of cropland expansion in the Brazilian Amazon. *Global Environmental Change*, 37, 92-101.

Munroe, R., Nathalie, D., Roe, D., Reid, H., Giuliani, A., Castelli, I., Möller, I. (2011). Does EbA work? A review of the evidence on the effectiveness of ecosystem-based approaches to adaptation. *IIED Policy Briefs*.

Murcia, C., Guariguata, M. R., Andrade, Á., Andrade, G. I., Aronson, J., Escobar, E. M., Etter, A., Moreno, F. H., Ramírez, W., & Montes, E. (2016). Challenges and prospects for scaling-up ecological restoration to meet international commitments: Colombia as a case study. *Conservation Letters*, 9(3), 213-220. https://doi.org/10.1111/conl.12199.

Natcher, D. C., & Hickey, C. G. (2002). Putting the community back into community-based resource management: a criteria and indicators approach to sustainability. *Human Organization*, 61(4), 350-363.

Naturevest & EKO (2014). *Investing in Conservation: A Landscape Assessment of an Emerging Market.* Natural Capital Investment Solutions and EKO Asset Management Partners. Available at: http://www.naturevesttnc.org/pdf/InvestingInConservation_Report.pdf

NCFA (2016). Available at: http://www.naturalcapitalfinancealliance.org/softcommoditytool/

Nehren, U., Thai, H. H. D., Marfai, M. A., Raedig, C., Alfonso, S., Sartohadi, J., & Castro, C. (2016). Ecosystem services of coastal dune systems for hazard mitigation: Case studies from Vietnam, Indonesia, and Chile. In F. G. Renaud, K. Sudmeier-Rieux, M. Estrella, U. Nehren. (Eds.). *Ecosystem-based disaster risk reduction and adaptation in practice* (pp. 401-434). Cham, Switzerland: Springer.

Nelson, A., & Chomitz, K. M. (2011). Effectiveness of strict vs. multiple use protected areas in reducing tropical forest fires: a global analysis using matching methods. *PLoS One*, 6(8), e22722. https://doi.org/10.1371/journal.pone.0022722.

Nemogá, G. R. (2005). *Regímenes de propiedad. In: Regímenes de Propiedad sobre Recursos Biológicos, Genéticos y Conocimiento Tradicional.* In G. R. Nemogá (Ed.). Series Plebio Nº 1. (pp. 27-44). Bogotá: Universidad Nacional de Colombia.

Nemogá, G., Chaparro, A., Pinto, L., Vallejo, F., Lizarazo, O., Rojas, D., Triana, V., Ávila, A., Blanco, J., Vanegas, P., & Jimenéz, O. (2010). Informe Final de la Propuesta de Ajuste al Régimen de Acceso a Recursos Genéticos y Productos Derivados, y a la Decisión Andina 391 de 1996. In G. Nemogá (Ed.). Series Plebio Nº 4. (pp. 63). Bogotá: Universidad Nacional de Colombia.

Nepstad, D., McGrath, D., Stickler, C., Alencar, A., Azevedo, A., Swette, B., Bezerra, T., Digiano, M., Shimada, J., Seroa da Motta, R., Armijo, E., Castello, L., Brando, P., Hansen, M. C., McGrath-Horn, Carvalho, M. O., & Hess, L. (2014). Slowing amazon deforestation through public policy and interventions in beef and soy supply chains. *Science*, 344, 1118-1123.

Newig, J., Challies, E., Jager, N. W., Kochskaemper, E., & Adzersen, A. (2017). The environmental performance of participatory and collaborative governance: a framework of causal mechanisms. *Policy Studies Journal*, 46(2), 269-297. https://doi.org/10.1111/psj.12209.

Nori, J., Lemes, P., Urbina-Cardona, N., Baldo, D., Lescano, J., Loyola, R. (2015). Amphibian conservation, land-use changes and protected areas: A global overview. *Biological Conservation*, 191, 367–374.

Nurse, L. A., McLean, R. F., Agard, J., Briguglio, L. P., Duvat-Magnan, V., Pelesikoti, N., Tompkins, E. L., & Webb, A. (2014). Small islands. In V. R. Barros, C. B. Field, D. J. Dokken, M. D. Mastrandrea, K. J. Mach, T. E. Bilir, M. Chatterjee, K. L. Ebi, Y. O. Estrada, R. C. Genova, B. Girma, E. S. Kissel, A. N. Levy, S. MacCracken, P. R. Mastrandrea, & L. L. White (Eds.). *Climate Change (2014): Impacts, Adaptation, and Vulnerability. Part B: Regional Aspects. Contribution of Working Group II to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change* (pp. 1613-1654). Cambridge, United Kingdom and New York, NY, USA: Cambridge University Press.

Nygren, A. (2005). Community-based forest management within the context of institutional decentralization in Honduras*. World Development*, 33(4), 639-655.

Ochieng, R. M., Visseren-Hamakers, I. J., Arts, B., Brockhaus, M. & Herold, M. (2016). Institutional effectiveness of REDD+ MRV: Countries progress in implementing technical guidelines and good governance. *Environmental Science & Policy*, 61, 42–52.

OECD (2016). *Active with Latin America and the Caribbean.* Paris, France: Organisation for Economic Co-operation and Development. Available at: http://www.oecd.org/globalrelations/Active-with-Latin-America-and-the-Caribbean.pdf

Oestreicher, J. S., Benessaiah, K., Ruiz-Jaen, M. C., Sloan, S., Turner, K., Pelletier, J., Guay, B., Clark, K. E., Roche, D. G., Meiners, M., & Potvin, C. (2009). Avoiding deforestation in Panamanian protected areas: an analysis of protection effectiveness and implications for reducing emissions from deforestation and forest degradation. *Global Environmental Change*, 19(2), 279-291.

Ojea, E., (2015). Challenges for mainstreaming ecosystem-based adaptation into the international climate agenda. *Current Opinion in Environmental Sustainability*, 14, 41-48.

Ölund Wingqvist, G. (2009). *Environmental and Climate Change in Latin America and the Caribbean - Policy Brief.* Sida-EEU, University of Gothenburg (pp. 27).

Ostrom, E. (1990). *Governing the Commons: the Evolution of Institutions for Collective Action.* New York: Cambridge University Press (pp. 280).

Ostrom, E. (2009). *A General Framework for Analyzing Sustainability of Social-Ecological Systems. Science*, 325, 419-422.

Ostrom, E., Dietz, T., Dolsak, N., Stern, P. C., Stonich, S., & Weber, E. U. (Eds.) (2002). *The Drama of the Commons*. Washington, DC: National Academy Press.

Ostrom, E., Janssen, M. A., & Anderies, J. M. (2007). Going beyond panaceas. *Proceedings of the National Academy of Sciences*, 104(39), 15176–15178.

Overbeck, G. E., Vélez-Martin, E., Scarano, F. R., Lewinsohn, T. M., Fonseca, C. R., Meyer, S. T., Müller, S. C., Ceotto, P., Dadalt, L., Durigan, G., Ganade, G., Gossner, M. M., Guadagnin, D. L., Lorenzen, K., Jacobi, C. M., Weisser, W. W., & Pillar, V. D. (2015). Conservation in Brazil needs to include non-forest ecosystems. *Diversity and Distributions*, 21, 1455-1460.

Pacheco, D. (2014). *Living-Well in Harmony and Balance with Mother Earth: a Proposal for Establishing a New Global Relationship Between Human Beings and Mother Earth*. La Paz: Universidad de la Cordillera. http://www.ucordillera.edu.bo/descarga/livingwell. pdf.

Palmer, M. A., & Ruhl, J. B. (2015). Aligning restoration science and the law to sustain ecological infrastructure for the future. *Frontiers in Ecology and the Environment*, 13(9), 512–519.

Pant, L. P., Adhikari, B., & Bhattarai, K. K. (2015). Adaptive transition for transformations to sustainability in developing countries. *Current Opinions in Environmental Sustainability,* 14, 206–212.

Pasteur, K., & Blauert, J. (2000). *Participatory Monitoring and Evaluation in Latin America: Overview of the Literature with Annotated Bibliography.* Development Bibliography, n.18. Brighton, England: Institute of Development Studies (pp. 81).

Pauli, H., Gottfried, M., Lamprecht, A., Niessner, S., Rumpf, S., Winkler, M., Steinbauer, K., & Grabherr, G. (Eds.) (2015). *The GLORIA Field Manual – Standard Multi-Summit Approach, Supplementary Methods and Extra Approaches*. 5th edition. GLORIA-Coordination. Vienna: Austrian Academy of Sciences & University of Natural Resources and Life Sciences. https://doi.org/10.2777/095439.

Pegas, F. V., & Buckley, R. (2012). Ecotourism (the Americas). In S. G. Beavis, M. L. Dougherty, & T. Gonzales (Eds.). *Berkshire Encyclopedia of Sustainability 8/10: The Americas and Oceania: Assessing Sustainability* (pp. 90-91). United States: Berkshire Publishing Group.

Pejchar, L., & Mooney, H. (2009). Invasive species, ecosystem services and human well-being. *Trends in Ecology and Evolution*, 24(9), 497-504.

Pereiro, X. (2016). A review of Indigenous tourism in Latin America: reflections on an anthropological study of Guna tourism (Panama). *Journal of Sustainable Tourism*, 24, 1121-1138.

Pérez-Ramírez, A., Castrejón, M., Gutiérrez, N. L., & Defeo, O. (2016), The Marine Stewardship Council certification in Latin America and the Caribbean: A review of experiences, potentials and pitfalls. *Fisheries Research*, 182, 50-58

Phelps, J., Friess, A. D., & Webb, E. L. (2012). Win–win REDD+ approaches belie carbon–biodiversity trade-offs. *Biological Conservation*, 154, 53–60.

Pinkerton, E., & Weinstein, M. (1995). *Fisheries that Work: Sustainability Through Community-Based Management*. A report to The David Suzuki Foundation. (pp. 212).

Pinto, S. R., Melo, F., Tabarelli, M., Padovesi, A., Mesquita, C. A., Scaramuzza, C. A. M., Castro, P., Carrascosa, H., Calmon, M., Rodrigues, R., César, R. G., & Brancalion, P. H. S. (2014). Governing and delivering a biome-wide restoration initiative: the case of Atlantic Forest Restoration Pact in Brazil. *Forests*, 5, 2212-2229.

Polisar, J., de Thoisy, B., Rumiz, D. I., Santos, F. D., McNab, R. B., Garcia-Anleu, R., Ponce-Santizo, G., Arispe, R., & Venegas, C. (2017). Using certified timber extraction to benefit jaguar and ecosystem conservation. *Ambio*, 46, 588-603.

Porter-Bolland, L., Ellis, E. A., Guariguata, M. R., Ruiz-Mallén, I., Negrete-Yankelevich, S., & Reyes-García, V. (2012). Community managed forests and forest protected areas: an assessment of their conservation effectiveness across the tropics. *Forest Ecology and Management*, 268, 6-17.

Porter-Bolland, L., Ruiz-Mallén, I., Camacho-Benavides, C., & McCandless, S. R. (Eds) (2013). *Community Action for Conservation - Mexican Experiences*. New York: Springer (pp. 184).

Posner, S., Getz, C., & Ricketts, T. (2016). Evaluating the impact of ecosystem service assessments on decision-makers. *Environmental Science & Policy*, 64, 30–37.

Pradhan, P., Costa, L., Rybski, D., Lucht, W., & Kropp, J. P. (2017). A systematic study of Sustainable Development Goal (SDG) interactions. *Earth's Future*, 5, 1169-1179. https://doi.org/10.1002/2017EF000632.

Pramova, E., Locatelli, B., Brockhaus, M., & Fohlmeister, S. (2012). Ecosystem services in the National Adaptation Programmes of Action. *Climate Policy*, 12, 393-409.

Pressey, R. L., Visconti, P., & Ferraro, P. J. (2015). Making parks make a difference: poor alignment of policy, planning and management with protected-area impact, and ways forward. *Philosophical Transactions of the Royal Society B Biological Sciences*, 370, 20140280.http://dx.doi.org/10.1098/rstb.2014.0280.

Preston, J. (2016). Canada. In D. Vinding & C. Mikkelsen (Eds.). *The Indigenous World 2016* (pp. 54-64). Copenhagen, Denmark: International Work Group for Indigenous Affairs.

Primack, R. B., Bray, D., Galletti, H. A., & Ponciano, I. (1998). *Timber, Tourists, and Temples: Conservation and Development in the Maya Forest of Belize, Guatemala and Mexico*. Washington: Island Press. (pp. 449).

PwC. (2010). *13th Annual Global CEO Survey 2010.* London: PricewaterhouseCoopers.

Queiroz, H. L. (2005). A Reserva de Desenvolvimento Sustentável Mamirauá*. Estudos Avançados,* 19(54), 183-203.

Queiroz, H. L., & Peralta, N. (2006). Reserva de Desenvolvimento Sustentável: Manejo integrado dos recursos naturais e gestão participativa. In I. Garay & B. Becker (Eds). *Dimensões Humanas da Biodiversidade* (pp. 447-476). Rio de Janeiro: Universidade Federal do Rio de Janeiro.

Quétier, F., & Lavorel, S. (2011). Assessing ecological equivalence in biodiversity offset schemes: Key issues and solutions. *Biological Conservation*, 144, 2991–2999.

Radachowsky, J., Ramos, V. H., McNab, R., Baur, E. H., & Kazakov, N. (2012). Forest concessions in the Maya Biosphere Reserve, Guatemala: a decade later. *Forest Ecology and Management*, 268, 18-28.

Redford, K. H., Huntley, B. J., Roe, D., Hammond, T., Zimsky, M., Lovejoy, T. E., Fonseca, G. A. B. da, Rodriguez, C. M., & Cowling, R. M. (2015). Mainstreaming biodiversity: conservation for the twenty-first century. *Frontiers in Ecology and Evolution*, 137, 1-7.

Reis, E. G., & D'Incao, F. (2000). The present status of artisanal fisheries of extreme Southern Brazil: an effort towards comminity-based management. *Ocean and Coastal Management*, 43, 585-595.

Renaud, F. G., Nehren. U., Sudmeier-Rieux, K., & Estrella, M. (2016). Developments and opportunities for ecosystem-based disaster risk reduction and climate change adaptation. In F. G. Renaud, K. Sudmeier-Rieux, M. Estrella, & U. Nehren (Eds.). *Ecosystem-Based Disaster Risk Reduction and Adaptation in Practice* (pp. 401-434). Cham, Switzerland: Springer.

Rhoads, B. L., Wilson, D., Urban, M., & Herricks, E. E. (1999). Interaction between scientists and nonscientists in community-based watershed management: emergence of the concept of stream naturalization. *Environmental Management*, 24(3), 297-308.

Ribeiro-Duthie, A. C., Domingos, L. M. B., Oliveira, M. F., Araujo, P. C., Alamino, R. C. J., Silva, R. S. V., Ribeiro-Duthie, J. M., & Castilhos, Z. C. (2017). Sustainable development opportunities within corporate social responsibility practices from LSM to ASM in the gold mining industry. *Mineral Economics*, 30, 141-152.

Richerzhagen, C. & Holm-Mueller, K. (2005). The effectiveness of access and benefit sharing in Costa Rica: Implications for national and international regimes. *Ecological Economics*, 53, 445-460.

Richerzhagen, C. (2011). Effective governance of access and benefit-sharing under the Convention on Biological Diversity. *Biodiversity and Conservation*, 20, 2243-2261.

Ricketts, T. H., Soares-Filho, B., da Fonseca G. A. B., Nepstad, D., Pfaff, A., Petsonk, A., Anderson, A., Boucher, D., Cattaneo, A., Conte, M., Creighton, K., Linden, L., Maretti, C., Moutinho, P., Ullman, R., Victurine, R. (2010): Indigenous lands, protected areas, and slowing climate change. *PLoS Biology*, 8(3), e1000331. https://doi.org/10.1371/journal.pbio. 1000331.

Ring, I., Hansjürgens, B., Elmqvist, T., Wittmer, H., & Sukhdev, P. (2010). Challenges in framing the economics of ecosystems and biodiversity: the TEEB initiative. *Current Opinion in Environmental Sustainability*, 2, 15–26.

Rissman, A. R., Owley, J., Shaw, M. R., & Thompson, B. (2015). Adapting conservation easements to climate change. *Conservation Letters*, 8, 68-76.

Rodrigues, A. S. L., Pilgrim, J. D., Lamoreux, J. F., Hoffmann, M., & Brooks, T. M. (2006). The value of the IUCN Red List for conservation. *Trends in Ecology and Evolution*, 21, 71–76.

Rodriguez Osuna, V. (2015). *Targeting watershed protection in the Guapiaçu-Macacu region of the Atlantic Forest, Brazil: An environmental and economic assessment of the potential for a payment for ecosystem services scheme.* Doctoral Thesis. Leipzig, Germany: Universität Leipzig.

Rodríguez Osuna, V., May, P., Monteiro, J., Wollenweber, R., Hissa, H., & Costa, M. (in press). Promoting sustainable agriculture, boosting productivity and enhancing climate mitigation and adaptation through the Rio Rural Program, Brazil. In U. Nehren, S. Schlueter, C. Raedig, D. Sattler, & H. Hissa (Eds.), *Strategies and tools for a sustainable Rio de Janeiro* (in press). Cham, Switzerland: Springer.

Rodríguez Osuna, V., Navarro Sánchez, G., Sommer, J. H., & Biber-Freudenberger, L. (2017). *Towards the Integration of Biodiversity in Environmental Impact Assessments of Bolivia*. Center for Development Research (ZEF), Universidad Católica Boliviana (UCB). Cochabamba-Bolivia: Editoria INIA.

Rodríguez, A. (2008). Tourism, indigenous peoples and conservation in the Ecuadorian Amazon. In A. Stronza & W. H. Durham (Eds.). *Ecotourism and Conservation in the Americas* (pp. 155-162).Wallingford, UK: Cabi International.

Roma, J. C., Saccaro, N. L., Mation, L. F., Paulsen, S. S., & Vasconcellos, P. G. (2013). *A Economia de Ecossistemas e da Biodiversidade no Brasil (TEEB-Brasil)*: *Análise de lacunas.* Rio de Janeiro: Instituto de Pesquisa Econômica Aplicada (IPEA) no. 1912.

Romero-Lankao, P., Gurney, K. R., Seto, K. C., Chester, M., Duren, R. M., Hughes, S., Hutyra, L. R., Marcotullio, P., Baker, L., Grimm, N. B., Kennedy, C., Larson, E., Pincetl, S., Runfola, D., Sanchez, L., Shrestha, G., Feddema, J., Sarzynski, A., Sperling, J., & Stokes, E. (2014). A critical knowledge pathway to low-carbon, sustainable futures: Integrated understanding of urbanization, urban areas, and carbon. *Earth's Future*, 2(10): 515-532. https://doi.org/10.1002/2014EF000258.

Rosado, A. & Craig, W. (2017). Biosafety regulatory systems overseeing the use of genetically modified organisms in the Latin America and Caribbean Region. *AgBioForum*, 20, 120-132.

Rudorff, B. F. T., Adami, M., Aguiar, D. A., Moreira, M. A., Mello, M. P., Fabiani, L., Amaral, D. F., & Pires, B. M. (2011). The Soy Moratorium in the Amazon Biome monitored by remote sensing images. *Remote Sensing*, 3, 185-202.

Rueda, X., & Lambin, E. F. (2013). Responding to globalization: impacts of certification on Colombian small-scale coffee growers. *Ecology and Society*, 18(3), p. art21. https://doi.org/10.5751/ES-05595-180321.

Sáenz, L., Mulligan, M., Arjona, F., & Gutierrez, T. (2014). The role of cloud forest restoration on energy security. *Ecosystem Services*, 9, 180-190.

Salas, S., Chuenpagdee, R., Seijo, J. C., & Charles, A. (2007). Challenges in the assessment and management of small-scale fisheries in Latin America and the Caribbean. *Fisheries Research*, 87, 5-16.

Santangeli, A., Toivonen, T., Pouzols, F. M., Pogson, M., Hastings, A., Smith, P., & Moilanen, A. (2016). Global change synergies and trade-offs between renewable energy and biodiversity. *Global Change Biology Bioenergy*, 8, 941-951.

Santini, L., Saura, S., & Rondinini, C. (2016). Connectivity of the global network of protected areas. *Diversity and Distributions*, 22, 199–211.

Sarkki, S., Niemelä, J., Tinch, R., van den Hove, S., Watt, A., & Young, J. (2014). Balancing credibility, relevance and legitimacy: A critical assessment of trade-offs in science–policy interfaces. *Science and Public Policy*, 41, 194-206.

Sarkki, S., Tinch, R., Niemelä, J., Heink, U., Waylen, K., Timaeus, J., Young, J., Watt, A., Neßhöver, C., & van den Hove, S. (2015). Adding 'iterativity' to the credibility, relevance, legitimacy: A novel scheme to highlight dynamic aspects of science–policy interfaces. *Environmental Science and Policy*, 54, 505-512.

Sattler, C., Schröter, B., Meyer, A., Giersch, G., Meyer, C., & Matzdorf, B. (2016). Multilevel governance in community-based environmental management: a case study comparison from Latin America. *Ecology and Society*, 21(4), 24. https://doi.org/10.5751/ES-08475-210424.

Sayer, J., & Campbell, B. (2004). *The Science of Sustainable Development: Local livelihoods and the Global Environment*. Cambridge: Cambridge University Press (pp. 290).

Sayles, J. S., & Baggio, J. A. (2017). Social–ecological network analysis of scale mismatches in estuary watershed restoration. *Proceedings of the National Academy of Sciences*, 114(10), E1776-E1785. https://doi.org/10.1073/pnas.1604405114.

Scarano, F. R. (2014). Plant conservation in Brazil: one hundred years in five. *Natureza e Conservação*, 12(1), 90-91.

Scarano, F. R. (2017). Ecosystem-based adaptation to climate change: concept, scalability and a role for conservation science. *Perspectives in Ecology and Conservation*, 15, 65-73.

Scarano, F. R., & Martinelli, G. (2010). Brazilian list of threatened plant species: reconciling scientific uncertainty and political decision-making. *Natureza e Conservação*, 8(1), 13-18.

Scarlett, L., & McKinney, M. (2016). Connecting people and places: the emerging role of network governance in large landscape conservation. *Frontiers in Ecology and the Environment,* 14(3), 116-125.

Schack, E., Beck, M., Brumbaugh, R., Crisley, K., & Hancock, B. (2012). *Restoration Works: Highlights from a Decade of Partnership Between The Nature Conservancy and the National Oceanic and Atmospheric Administration's Restoration Center*. Arlington, USA: The Nature Conservancy.

Schaefer, M., Goldman, E., Bartuska, A. M., Sutton-Grier, A., & Lubchenco, J. (2015). Nature as capital: Advancing and incorporating ecosystem services in United States federal policies and programs. *Proceedings of the National Academy of Sciences*, 112, 7383-7389.

Schafer, A. G., & Reis, E. G. (2008): Artisanal fishing areas and traditional ecological knowledge: the case study of the artisanal fisheries of the Patos Lagoon estuary (Brazil). *Marine Policy*, 32, 283-292.

Schipper, J., Chanson, J. S., Chiozza F., Cox, N. A., Hoffmann, M., Katariya, V., Lamoreux, J., Rodrigues, A. S. L., Stuart, S. N., Temple, H. J., Baillie, J., Boitani, L., Lacher Jr., T. E., Mittermeier, R. A., Smith, A. T., Absolon, D., Aguiar, J. M., Amori, G., Bakkour, N., Baldi, R., Berridge, R. J., Bielby, J., Black, P. A., Blanc, J. J., Brooks, T. M., Burton, J. A. Butynski, T. M. Catullo, G., Chapman, R., Cokeliss, Z., Collen, B., Conroy, J., Cooke, J. G., da Fonseca, G. A. B., Derocher, A. E., Dublin, H. T., Duckworth, J. W., Emmons, L., Emslie, R. H., Festa-Bianchet, M., Foster, M., Foster, S., Garshelis, D. L., Gates, C., Gimenez-Dixon, M., Gonzalez, S., Gonzalez-Maya, J. F., Good, T. C., Hammerson, G., Hammond, P. S., Happold, D., Happold, M., Hare, J., Harris, R. B., Hawkins, C. E., Haywood, M., Heaney, L. R., Hedges, S., Helgen, K. M., Hilton-Taylor, C., Hussain, S. A., Ishii, N., Jefferson, T. A., Jenkins, R. K. B., Johnston, C. H., Keith, M., Kingdon, J., Knox, D. H., Kovacs, K. M., Langhammer, P., Leus, K., Lewison, R., Lichtenstein, G., Lowry, L. F., Macavoy, Z., Mace, G. M., Mallon, D. P., Masi, M., McKnight, M. W., Medellín, R. A., Medici, P., Mills, G., Moehlman, P. D., Molur, S., Mora, A., Nowell, K., Oates, J. F., Olech, W., Oliver, W. R. L., Oprea, M., Patterson, B. D., Perrin, W. F., Polidoro, B. A., Pollock, C., Powel, A., Protas, Y., Racey, P., Ragle, J., Ramani, P., Rathbun, G., Reeves, R. R., Reilly, S. B., Reynolds III, J. E., Rondinini, C., Rosell-Ambal, R. G., Rulli, M., Rylands, A. B., Savini, S., Schank, C. J., Sechrest, W., Self-Sullivan, C., Shoemaker, A., Sillero-Zubiri, C., De Silva, N., Smith, D. E., Srinivasulu, C., Stephenson, P. J., van Strien, N., Talukdar, B. K., Taylor, B. L., Timmins, R., Tirira, D. G., Tognelli, M. F., Tsytsulina, K., Veiga, L. M., Vié, J. –C., Williamson, E. A., Wyatt, S. A., Xie, Y., & Young, B. E. (2008). The status of the world's land and marine mammals: diversity, threat, and knowledge. *Science*, 322, 225–230.

Schöngart, J., & Queiroz, H. L. (2010). Traditional timber harvesting in the Central Amazonian Floodplains. In W. J. Junk, M. T. F. Piedade, F. Wittmann, J. Schöngart, & P. Parolin (Eds.). *Amazonian Floodplain Forests - Ecophysiology, Biodiversity and Sustainable Management*. Ecological Studies Vol. 20. (pp. 419-436). New York: Springer.

Seddon, A. W. R., Macias-Fauria, M., Long, P. R., Benz, D., & Willis, K. J. (2016). Sensitivity of global terrestrial ecosystems to climate variability. *Nature*, 531, 229-232.

Segan, D. B., Murray, K. A., & Watson, J. E. M. (2016). A global assessment of current and future biodiversity vulnerability to habitat loss–climate change interactions. *Global Ecology and Conservation*, 5, 12–21.

Sienra, A. M. M., & Medina, C. E. (2013). Reservas naturales en el Paraguay. Aporte del sector privado a la conservación de la biodiversidad. In: F. Arano & J. Egea (Eds.). *Conjugando Producción y Conservación en el Chaco Paraguayo* (pp. 24-29). Asunción, Paraguay: WCS-AVINA. (pp. 60).

Silva, N. P., & Brancher, P. T. (2014). Economia e política externa: um balanço do governo Lula (2002/2010). *Revista Perspectivas do Desenvolvimento: um Enfoque Multidimensional 2*(3), 1-23.

Silvano, R. A. M., Hallwass, G., Lopes, P. F., Ribeiro, A. R., Lima, R. P., Hasenack, H., Jura, A. A., & Begossi, A. (2014). Co-management and spatial features contribute to secure fish abundance and fishing yields in tropical floodplain lakes. *Ecosystems*, 17, 271-285.

Silvestri, L. (2014). *La obligación de participación justa y equitativa en los beneficios derivados de la utilización de recursos genéticos establecida en el protocolo de Nagoya: ¿una oportunidad de desarrollo para países ricos en biodiversidad?* Doctoral thesis, Madrid, España: Universidad Rey Juan Carlos (pp. 523).

Silvestri, L., (2015): *La conservación de la diversidad genética argentina: tres desafíos para implementar el régimen de acceso a los recursos genéticos y la distribución de los beneficios. Revista Ecología Austral*, 25, 273 -278.

Simberloff, D. (2011). *How common are invasion-induced ecosystem impacts? Biological Invasions*, 13, 1255–1268.

Singh, S. P., & Singh, A. M. (2015). Bioprospecting: enhancing the value of biodiversity and intellectual property. In R. K. Salgotra & B. B. Gupta (Eds.). *Plant genetic resources and traditional knowledge for food security* (pp. 231-236). Singapore: Springer.

Skirycz, A., Kierszniowska, S., Méret, M., Willmitzer, L., & Tzotzos, G. (2016). Medicinal bioprospecting of the Amazon Rainforest: a modern Eldorado? *Trends in Biotechnology*, 34, 781-790.

Slowley, G. A. (2001). Globalization and self-government: impacts and implications for First Nations in Canada. *The American Review of Canadian Studies*, 31(1-2), 265-281.

Snäll, T., Lehtomäki, J., Arponen, A., Elith, J., & Moilanen, A. (2016). Green infrastructure design based on spatial conservation prioritization and modeling of biodiversity features and ecosystem services. *Environmental Management*, 57, 251–256.

Soares-Filho, B., Rajão, R., Macedo, M., Carneiro, A., Costa, W., Coe, M., Rodrigues, H., & Alencar, A. (2014). Cracking Brazil's Forest Code. *Science*, 344, 363-364.

Soares-Filho, B., Rajão, R., Merry, F., Rodrigues, H., Davis, J., Lima, L., Macedo, M., Coe, M., Carneiro, A., & Santiago, L. (2016). Brazil's market for trading forest certificates. *PLoS One*, 11, e0152311. https://doi.org/10.1371/journal.pone.0152311

Soriano, M., Kainer, K. A., Staudhammer, C. L., & Soriano, E. (2012). Implementing multiple forest management in Brazil nut-rich community forests: effects of logging on natural regeneration and forest disturbance. *Forest Ecology and Management*, 268, 92-102.

Southgate, D., Haab, T., Lundine, J., & Rodríguez, F. (2009). Payments for environmental services and rural livelihood strategies in Ecuador and Guatemala. *Environment and Development Economics*, 15, 21–37.

Souza, P. R., & Queiroz, H. L. (2008). A participação dos aruanãs (*Osteoglossum bicirrhosum*) nos ilícitos registrados pelo sistema de fiscalização da Reserva Mamirauá. In: H. L. Queiroz & M. Camargo (Eds.). **Biologia, Conservação e Manejo dos Aruanãs na Amazônia Brasileira** (p. 41-60). Tefé, Brasil: Instituto de Desenvolvimento Sustentável Mamirauá.

Souza, V. L. P. (2015). Integração territorial no Mercosul: o caso da IIRSA/COSIPLAN. *Sociedade e Natureza*, 27(1), 21-35.

Soysal, M., Bloemhof-Ruwaard, J. M., & van der Vorst, J. G. A. J. (2014). Modelling food logistics networks with emission considerations: The case of an international beef supply chain. *International Journal of Production Economics*, 152, 57–70.

Spangenberg, J. H., & Settele, J. (2010). Precisely incorrect? Monetising the value of ecosystem services. *Ecological Complexity*, 7, 327-337.

Speziale, K. L., Lambertucci, S. A., Carrete, M., & Tella, J. L. (2012). Dealing with non-native species: what makes the difference in South America? *Biological Invasions*, 14, 1609–1621.

Steffen, W., Broadgate, W., Deutsch, L., Gaffney, O., Ludwig, C. (2015a). The trajectory of the Anthropocene: the Great Acceleration. *The Anthropocene Review*, 2, 81-98.

Steffen, W., Richardson, K., Rockström, J., Cornell, S. E., Fetzer, I., Bennett, E. M., Biggs, R., Carpenter, S. R., de Vries, W., de Wit, C. A., Folke, C., Gerten, D., Heinke, J., Mace, G. M., Persson, L. M., Ramanathan, V., Reyers, B., & Sörlin, S. (2015b). Planetary boundaries: guiding human development on a changing planet. *Science*, 347, 1259855. https://doi.org/10.1126/science.1259855.

Stephenson, P. J., Brooks, T. M., Butchart, S. H., Fegraus, E., Geller, G. N., Hoft, R., Hutton, J., Kingston, N., Long, B., & McRae, L. (2017). Priorities for big biodiversity data. *Frontiers in Ecology and the Environment*, 15(3), 124-125.

Stockholm University. (2014). *Applying resilience thinking: Seven principles for building resilience in social-ecological systems.* Stockholm Resilience Centre and Stockholm University, 20 p.

Stronza, A. (2008). The bold agenda of ecotourism. In A. Stronza & W. H. Durham (Eds.). *Ecotourism and Conservation in the Americas* (pp. 3-17). Wallingford, UK: Cabi International.

Tayleur, C., Balmford, A., Buchanan, G. M., Butchart, S. H. M., Ducharme, H., Green, R. E., Milder, J. C., Sanderson, F. J., Thomas, D. H. L., Vickery, J., & Phalan, B. (2017). Global coverage of agricultural sustainability standards, and their role in conserving biodiversity. *Conservation Letters*, 10(5), 610-618. https://doi.org/10.1111/conl.12314.

TEEB (2010). *The Economics of Ecosystems and Biodiversity Ecological and Economic Foundations.* London and Washington: Earthscan.

Ten Kate, K., & Wells, A. 2001. *Preparing a National Strategy on Access to Genetic Resources and Benefit-Sharing: A Pilot Study*. Richmond, UK: Royal Botanic Gardens Kew (74 pp).

Traxler, G. (2006). The GMO experience in North and South America. *International Journal of Technology and Globalisation,* 2 (1/2), 46-64.

Trejo-Salazar, R. -E., Eguiarte, L. E., Suro-Piñera, D., & Medellin, R. A. (2016). Save our bats, save our tequila: Industry and science join forces to help bats and agaves. *Natural Areas Journal*, 36(4), 523–530. https://doi.org/10.3375/043.036.0417

Trinidad and Tobago (2016): *Fifth National Report to the United Nations Convention on Biological Diversity. Government of the Republic of Trinidad & Tobago.* Port of Spain, Trinidad. Available at: https://www.cbd.int/doc/world/tt/tt-nr-05-en.pdf.

Triyanti, A., & Chu, E. K. (2017). A survey of governance approaches to ecosystem-based disaster risk reduction: Current gaps and future directions. *International Journal of Disaster Risk Reduction* (in press). https://doi.org/10.1016/j.ijdrr.2017.11.005.

Tucker, C. M. (2010). Learning on governance in forest ecosystems: Lessons from recent research. *International Journal of the Commons*, 4(2), 687–706.

UN (2015a). *MDG Success Springboard for New Sustainable Development Agenda: United Nations Report. Latin America and the Caribbean Continue to Show Impressive Gains in Many Areas, Although Disparities Exist Between the Sub-Regions*. Available at www.un.org/millenniumgoals/2015_MDG_Report/pdf/MDG%202015%20PR%20Regional%20LAC.pdf

UN (2015b). *Transforming Our World: The 2030 Agenda for Sustainable Development. United Nations*. Available at: sustainabledevelopment.un.org

UN (2016). *Sustainable Development Knowledge Platform. United Nations, Department of Economic and Social Affairs*. Available at: https://sustainabledevelopment.un.org

UNDP (2015). *2015 Human Development Report. United Nations Development Program, New York.*

UNEP (2012). *GEO-5: Environment for the Future We Want.* Nairobi, Kenya: United Nations Environment Programme.

UNEP (2013). *South-South Cooperation: Sharing National Pathways Towards Inclusive Green Economies.* Geneva, Switzerland: United Nations Environment Programme.

UNEP Finance Initiative (2010). *Demystifying Materiality: Hardwiring Biodiversity and Ecosystem Services Into Finance. CEO Briefing.* Geneva, Switzerland: United Nations Environment Programme. Available at: http://www.unepfi.org/fileadmin/documents/CEO_ DemystifyingMateriality.pdf

UNEP Finance Initiative (2015). *Bank and Investor Risk Policies on Soft Commodities.* Geneva, Switzerland: United Nations Environment Programme. Available at: http://www.naturalcapitalfinancealliance.org/documents/wgi/NCD%20-%20SOFT%20 COMMODITIES%20RISK%20(FULL).pdf.

UNEP-WCMC (2016). *The State of Biodiversity in Latin America and the Caribbean: A Mid-Term Review of Progress Towards the Aichi Biodiversity Targets.* Cambridge, UK: UNEP-WCMC.

UNEP-WCMC, & IUCN (2016). *Protected Planet Report 2016.* Cambridge UK and Gland, Switzerland: UNEP-WCMC and IUCN.
UNEP-WCMC, & IUCN (2017). *Protected Planet Report 2017.* Cambridge UK and Gland, Switzerland: UNEP-WCMC and IUCN.
van Heerwaarden, J., Hellin, J., Visser, R. F., & van Eeuwijk, F. A. (2009). Estimating maize genetic erosion in modernized smallholder agriculture. *Theoretical Applied Genetics,* 119, 875-888.
VanWey, L. K., & Richards, P. D. (2014). Eco-certification and greening the Brazilian soy and corn supply chains. *Environmental Research Letters*, 9(3), 31002. https://doi.org/10.1088/1748-9326/9/3/031002.
Vargas Roncancio, I. D., & Nemogá Soto, G. R. (2010). Contrato de acceso a recursos genéticos: Un análisis comparado. *Revista Pensamiento Jurídico*, 27, 157-202.
Vasseur, L., Horning, D., Thornbush, M., Cohen-Shacham, E., Andrade, A., Barrow, E., Edwards, S. R., Wit, P., & Jones, M. (2017). Complex problems and unchallenged solutions: Bringing ecosystem governance to the forefront of the UN sustainable development goals. *Ambio*, 46, 731–742.
Venter, O., (2014). Corridors of carbon and biodiversity. *Nature Climate Change*, 4, 91-92.
Vignola, R., Harvey, C. A., Bautista-Solis, P., Avelino, J., Rapidel, B., Donatti, C., & Martinez, R. (2015). Ecosystem-based adaptation for smallholder farmers: definitions, opportunities and constraints. *Agriculture, Ecosystems and Environment*, 211, 126–132.
Villareal, M. A., & Fergusson, I. F. (2017). *The North American Free Trade Agreement (NAFTA)*. CRS Report R42965. Washington, D.C.: Congressional Research Service.
Villarroya, A., Barros, A. C., & Kiesecker, J. (2014). Policy development for environmental licensing and biodiversity offsets in Latin America. *PLoS One*, 9(9), e107144. https://doi.org/10.1371/journal.pone.0107144
Vinding, D., & Jensen, M. W. (2016). Editorial. In D. Vinding, & C. Mikkelsen (Eds.). *The Indigenous World 2016* (pp. 10-21). Copenhagen, Denmark: International Work Group for Indigenous Affairs.
Visconti, P., Bakkenes, M., Baisero, D., Brooks, T., Butchart, S. H. M., Joppa, L., Alkemade, R., Di Marco, M., Santini, L., Hoffmann, M., Maiorano, L., Pressey, R. L., Arponen, A., Boitani, L., Reside, A. E., van Vuuren, D. P., & Rondinini, C. (2015). Projecting global biodiversity indicators under future development scenarios. *Conservation Letters*, 9(1), 5-13.
Vitel, C. S. M. N., Carrero, G. C., Cenamo, M. C., Leroy, M., Graça, P. M. L. A., & Fearnside, P. M. (2013). Land-use change modeling in a Brazilian indigenous reserve: Construction of a reference scenario for the Suruí REDD project. *Human Ecology*, 41(6), 807-826.
Vitte, C. C. S. (2011). *Integração da Infraestrutura Produtiva e Ordenamento Territorial na América do Sul: Geoeconomia e Política Externa Brasileira Como Condicionantes.* XIV Encontro Nacional da ANPUR, Maio de 2011 (pp. 1-21). Rio de Janeiro, Brasil.
Von Stechow, C., Minx, J. C., Riahi, K., Jewell, J., McCollum, D. L., Callaghan, M. W., Bertram, C., Luderer, G., & Baiocchi, G. (2016). 2 °C and SDGs: united they stand, divided they fall? *Environmental Research Letters*, 11, 034022. https://doi.org/10.1088/1748-9326/11/3/034022.
Waite, R., Kushner, B., Jungwiwattanaporn, M., Gray, E., & Burke, L. (2015). Use of coastal economic valuation in decision making in the Caribbean: Enabling conditions and lessons learned. *Ecosystem Services*, 11, 45-55.
Walker, B., & Salt, D. (2012). Resilience Practice: Building Capacity to Absorb Disturbance and Maintain Function. Washington, DC.: Island Press.
Walsh-Dilley, M., Wolford, W., & McCarthy, J. (2016). Rights for resilience: food sovereignty, power, and resilience in development practice. *Ecology and Society*, 21(1), 11. http://dx.doi.org/10.5751/ES-07981-210111.
Watson, J. E. M., Dudley, N., Segan, D. B., & Hockings, M. (2014). The performance and potential of protected areas. *Nature*, 515, 67-73.
WBCSD (2017). *Sustainability and Enterprise Risk Management: The First Step Towards integration.* Geneva: World Business Council on Sustainable Development.
WEF (2017). Commodities and Forests Agenda 2020: Ten priorities to remove tropical deforestation from commodity supply chains. World Economic Forum, Geneva. Available at

https://www.tfa2020.org/wp-content/uploads/2017/09/TFA2020_CommoditiesandForestsAgenda2020_Sept2017.pdf

WEF, (2015): *Global Risks 2015.* 10th edition. Geneva: World Economic Forum. Available at: http://www3.weforum.org/docs/WEF_Global_Risks_2015_Report15.pdf

Weichselgartner, J., & Kasperson, R. (2010). Barriers in the science-policy-practice interface: Toward a knowledge-action-system in global environmental change research. *Global Environmental Change*, 20, 266-277.

Weiss, M., & Cattaneo, C. (2017). Degrowth – Taking stock and reviewing an emerging academic paradigm. *Ecological Economics*, 137, 220–230.

Whitford, M., & Ruhanen, L. (2016). Indigenous tourism research, past and present: where to from here? *Journal of Sustainable Tourism*, 24, 1080-1099.

Wiber, M., Berkes, F., Charles, A., & Kearney, J. (2004). Participation research supporting community-based fishery management. *Marine Policy*, 28, 459-468.

Williams, P. W., & Peters, M. (2008). Entrepreneurial performance and challenges for aboriginal small tourism businesses: A Canadian case tourism. *Recreation Research*, 33(3), 277–287.

Witkowski, K., & Medina, D. (2016). *El Sector Agropecuario en las Contribuciones Previstas y Determinadas a Nivel Nacional de América Latina*. Instituto Interamericano de Cooperación para la Agricultura (IICA) (28p).

Witkowski, K., Medina, D., & Garcia, M. (2016). *Intended Nationally Determined Contributions in the Caribbean: Where Does Agriculture Fit?* Instituto Interamericano de Cooperación para la Agricultura (IICA) (28p).

Woosnam, K. M., & Kim, H. (2014). Hurricane impacts on southeastern United States coastal national park visitation. *Tourism Geographies*, 16, 364-381.

World Bank (2012). The World Bank Data. In *World Development Indicators, Urban Development, Urban Population (% of Total) and Population in the Largest City (% of Urban Population).* Washington, DC, USA: The World Bank. Available at: worldbank.org/topic/urban-development

WTTC (2016). *The Comparative Economic Impact of Travel and Tourism.* London, UK. World Travel and Tourism Council. Available at http://wttc.org

Wünscher, T., Engel, S., & Wunder, S. (2008). Spatial targeting of payments for environmental services: A tool for boosting conservation benefits. *Ecological Economics*, 65, 822-833.

Zachos, F. E., & Habel, J. C. (Eds.) (2011). *Biodiversity Hotspots: Distribution and Protection of Conservation Priority Areas*. Heidelberg, Dordrecht, London, New York: Springer.

Zanella, M. A., & Milhorance, C. (2016). Cerrado meets savannah, family farmers meet peasants: The political economy of Brazil's agricultural cooperation with Mozambique. *Food Policy*, 58, 70-81.

Zanella, M. A., Schleyer, C., & Speelman, S. (2014). Why do farmers join Payments for Ecosystem Services (PES) schemes? An assessment of PES water scheme participation in Brazil. *Ecological Economics*, 105, 166-176.

Zimmermann, E., Bracalenti, L., Piacentini, R., & Inostroza, L. (2016). Urban flood risk reduction by increasing green areas for adaptation to climate change. *Procedia Engineering*, 161, 2241-2246.

Annex I - Glossary

A

Abundance: the size of a population of a particular life form in a given area.
acidification Ongoing decrease in pH away from neutral value of 7. Often used in reference to oceans, freshwater or soils, as a result of uptake of carbon dioxide from the atmosphere.

Access and benefit sharing (ABS): One of the three objectives of the Convention on Biological Diversity, as set out in its Article 1, is the "fair and equitable sharing of the benefits arising out of the utilization of genetic resources, including by appropriate access to genetic resources and by appropriate transfer of relevant technologies, taking into account all rights over those resources and to technologies, and by appropriate funding". The CBD also has several articles (especially Article 15) regarding international aspects of access to genetic resources.

Adaptation: adjustment in natural or human systems to a new or changing environment, whether through genetic or behavioural change.
adaptive capacity The general ability of institutions, systems, and individuals to adjust to potential damage, to take advantage of opportunities, or to cope with the consequences.

Adaptive management: a systematic process for continually improving management policies and practices by learning from the outcomes of previously employed policies and practices. In active adaptive management, management is treated as a deliberate experiment for purposes of learning.

Afforestation: converting grasslands or shrublands into tree plantations. Afforestation is sometimes suggested as a tool to sequester carbon, but it can have negative impacts on biodiversity and ecosystem function.

Agenda setting: one of four phases in the policy cycle. Agenda setting motivates and sets the direction for policy design and implementation.

Agricultural intensification: an increase in agricultural production per unit of inputs (which may be labour, land, time, fertilizer, seed, feed or cash).

Agrobiodiversity: agricultural biodiversity is the biological diversity that sustains key functions, structures and processes of agricultural ecosystems. It includes the variety and variability of animals, plants and micro-organisms, at the genetic, species and ecosystem levels.

Agro-ecological zones: geographic areas with homogeneous sets of climatic parameters and natural resource characteristics, such as rainfall, solar radiation, soil types and soil qualities, which correspond to a level of agricultural potential.

Agroecology: the science and practice of applying ecological concepts, principles and knowledge (i.e., the interactions of, and explanations for, the diversity, abundance and activities of organisms) to the study, design and management of sustainable agroecosystems. It includes the roles of human beings as a central organism in agroecology by way of social and economic processes in farming systems. Agroecology examines the roles and interactions among all relevant biophysical, technical and socioeconomic components of farming systems and their surrounding landscapes.

Agroecosystem: an ecosystem, dominated by agriculture, containing assets and functions such as biodiversity, ecological succession and food webs. An agroecosystem is not restricted to the immediate site of agricultural activity (e.g. the farm), but rather includes the region that is impacted by this activity, usually by changes to the complexity of species assemblages and energy flows, as well as to the net nutrient balance.

Agroforestry: a collective name for land-use systems and technologies where woody perennials (trees, shrubs, palms, bamboos, etc.) are deliberately used on the same land-management units as agricultural crops and animals, in some form of spatial arrangement or temporal sequence.

Aichi (Biodiversity) Targets: the 20 targets set by the Conference of the Parties to the Convention for Biological Diversity (CBD) at its tenth meeting, under the Strategic Plan for Biodiversity 2011-2020.
Alien species See "invasive alien species".

Annual: in botany, refers to plants that grow from seed to maturity, reproduction and death in one year. Related terms are biennial (plants that take two years to complete their life cycles), and perennial (plants that take several many years to complete their life cycles).

Anthropogenic assets: built-up infrastructure, health facilities, or knowledge - including indigenous and local knowledge systems and technical or scientific knowledge - as well as formal and non-formal education, work, technology (both physical objects and procedures), and financial assets. Anthropogenic assets have been highlighted to emphasize that a good quality of life is achieved by a co-production of benefits between nature and people.

Anthropogenic impact: Impacts resulting from human activities.

Approval: approval of the Platform's outputs signifies that the material has been subject to detailed, line-by-line discussion and agreement by consensus at a session of the Plenary.

Aquaculture: the farming of aquatic organisms, including fish, molluscs, crustaceans and aquatic plants, involving interventions such as regular stocking, feeding, protection from predators, to enhance production. (In contrast, aquatic organisms which are exploitable by the public as a common property resource, are classed as fisheries, not aquaculture).

Archetypes: in the context of scenarios, an over-arching scenario that embodies common characteristics of a number of more specific scenarios.
Arid ecosystems Those in which water availability severely constrains ecological activity.

Assessment reports: published outputs of scientific, technical and socioeconomic issues that take into account different approaches, visions and knowledge systems, including global assessments of biodiversity and ecosystem services with a defined geographical scope, and thematic or methodological assessments based on the standard or the fast-track approach. They are to be composed of two or more sections including a summary for policymakers, an optional technical summary and individual chapters and their executive summaries. Assessments are the major output of IPBES, and they contain syntheses of findings on topics that have been selected by the IPBES Plenary.

B

Backcasting: an analytical technique used to search for target-seeking scenarios that fulfil a predefined goal, or set of goals.

Baseline: a minimum or starting point with which to compare other information (e.g. for comparisons between past and present or before and after an intervention).

Benefit sharing: distribution of benefits between stakeholders.

Benefits: advantage that contributes to wellbeing from the fulfilment of needs and wants. In the context of nature's contributions to people (see "Nature's contributions to people"), a benefit is a positive contribution. (There may also be negative contributions, dis-benefits, or costs, from Nature, such as diseases).

Benthic: occurring at the bottom of a body of water; related to benthos.

Benthos: a group of organisms, other invertebrates, that live in or on the bottom in aquatic habitats.

Bioaccumulation: Some contaminants that enter biological systems are preferentially stored (usually in fat tissue) in organisms resulting in an accumulation over time. This process is called bioaccumulation.

Biocapacity: the ecosystem's capacity to produce biological materials used by people and to absorb waste material generated by humans, under current management schemes and extraction technologies". The "biocapacity" indicator used in the present report is based on the Global Footprint Network, unless otherwise specified.

Biodiversity: the variability among living organisms from all sources including terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are a part. This includes variation in genetic, phenotypic, phylogenetic, and functional attributes, as well as changes in abundance and distribution over time and space within and among species, biological communities and ecosystems.

Biodiversity footprint hotspots: biodiversity threat hotspots driven by global consumption of goods and services.

Biodiversity hotspot: a generic term for an area high in such biodiversity attributes as species richness or endemism. It may also be used in assessments as a precise term applied to geographic areas defined according to two criteria (Myers et al 2000): (i) containing at least 1,500 species of the world's 300,000 vascular plant species as endemics, and (ii) being under threat, in having lost 70 % of its primary vegetation.

Biodiversity loss: the reduction of any aspect of biological diversity (i.e. diversity at the genetic, species and ecosystem levels) is lost in a particular area through death (including extinction), destruction or manual removal; it can refer to many scales, from global extinctions to population extinctions, resulting in decreased total diversity at the same scale.

Biodiversity offset: a biodiversity offset is a tool proposed by developers and planners for compensating for the loss of biodiversity in one place by biodiversity gains in another.

Biofuel: fuel made from biomass.

Biomass: the mass of non-fossilized and biodegradable organic material originating from plants, animals and micro-organisms in a given area or volume.

Biome: global-scale zones, generally defined by the type of plant life that they support in response to average rainfall and temperature patterns. For example, tundra, coral reefs or savannas.

Biosphere: the sum of all the ecosystems of the world. It is both the collection of organisms living on the Earth and the space that they occupy on part of the Earth's crust (the lithosphere), in the oceans (the hydrosphere) and in the atmosphere. The biosphere is all the planet's ecosystems.

Biota: all living organisms of an area; the flora and fauna considered as a unit.

Bonn Challenge: a global effort to restore 150 million hectares of the world's degraded and deforested lands by 2020 and 350 million hectares by 2030. It is overseen by the Global Partnership on Forest Landscape Restoration, with the International Union for Conservation of Nature as its Secretariat.

Boundary objects: objects and/or processes plastic enough to adapt to local needs and to the constraints of the several parties employing them, yet robust enough to maintain a common identity across sites. Their meanings may differ in different social contexts, but their structure is common enough and recognizable across contexts.

Buen vivir: although no universal definition of buen vivir has been attained yet, it has “four common constitutive elements: (a) the idea of harmony with nature (including its abiotic components); (b) vindication of the principles and values of marginalized/subordinated peoples; (c) the State as guarantor of the satisfaction of basic needs (such as education, health, food and water), social justice and equality; and (d) democracy. There are also two cross-cutting lines: buen vivir as a critical paradigm of Eurocentric (anthropocentric, capitalist, economistic and universalistic) modernity, and as a new intercultural political project”.

Bushmeat: meat for human consumption derived from wild animals.

Bycatch: the commercially undesirable species caught during a fishing process.

C

Capacity-building (or development): defined by the United Nations Development Programme as “the process through which individuals, organisations and societies obtain, strengthen and maintain their capabilities to set and achieve their own development objectives over time”. IPBES promotes and facilitates capacity-building, to improve the capacity of countries to make informed policy decisions on biodiversity and ecosystem services.

Carbon cycle: the carbon cycle is the process by which carbon is exchanged among the ecosystems of the Earth.

Carbon footprint: a measure of the total amount of carbon dioxide emissions, including carbon dioxide equivalents, that is directly and indirectly caused by an activity or is accumulated over the life stages of a product.

Carbon sequestration: the long-term storage of carbon in plants, soils, geologic formations, and the ocean. Carbon sequestration occurs both naturally and as a result of anthropogenic activities and typically refers to the storage of carbon that has the immediate potential to become carbon dioxide gas.

Carbon storage: the biological process by which carbon in the form carbon dioxide is taken up from the atmosphere and incorporated through photosynthesis into different compartments of ecosystems, such as biomass, wood, or soil organic carbon. Also, the technological process of capturing waste carbon dioxide from industry or power generation, and storing it so that it will not enter the atmosphere.

Carrying capacity: in ecology, the carrying capacity of a species in an environment is the maximum population size of the species that the environment can sustain indefinitely. The term is also used more generally to refer to the upper limit of habitats, ecosystems, landscapes, waterscapes or seascapes to provide tangible and intangible goods and services (including aesthetic and spiritual services) in a sustainable way.

Certainty: in the context of IPBES, the summary terms to describe the state of knowledge are the following:

- Well established (Certainty term (q.v.)): comprehensive meta-analysis or other synthesis or multiple independent studies that agree.
- Established but incomplete (Certainty term (q.v.)): general agreement although only a limited number of studies exist but no comprehensive synthesis and, or the studies that exist imprecisely address the question.
- Unresolved (Certainty term (q.v.)): multiple independent studies exist but conclusions do not agree.
- Inconclusive (Certainty term (q.v.)): limited evidence, recognising major knowledge gaps.

Climate change: as defined in Article 1 of the UNFCCC, "a change of climate which is attributed directly or indirectly to human activity that alters the composition of the global atmosphere and which is in addition to natural climate variability observed over comparable time periods".

Co-management: process of management in which government shares power with resource users, with each given specific rights and responsibilities relating to information and decision-making.

Co-production: in the context of the IPBES conceptual framework, this is the joint contribution by nature and anthropogenic assets in generating nature's contributions to people.

Compensation: a given project attains zero net biodiversity loss when its unavoidable impacts on biodiversity are balanced out or compensated by actions such as conservation, rehabilitation, restoration and/or compensation of residual impacts that avoid or minimize losses. In this case, compensation refers to environmental compensation and not socioeconomic compensation to the people who are affected by the project's impact.

Conservation easement: voluntary, typically permanent, partial interest in property created through agreement between a landowner and a nonprofit land trust or government agency in which a landowner agrees to land-use restrictions, usually in exchange for a payment, tax reduction, or permit.

Corridor: a geographically defined area which allows species to move between landscapes, ecosystems and habitats, natural or modified, and ensures the maintenance of biodiversity and ecological and evolutionary processes.

Cropland: a land cover/use category that includes areas used for the production of crops for harvest.

Cross-scale analysis: cross-scale effects are the result of spatial and/or temporal processes interacting with other processes at another scale. These interactions create emergent effects that can be difficult to predict.

D

Decomposition: breakdown of complex organic substances into simpler molecules or ions by physical, chemical and/or biological processes

Deforestation: human-induced conversion of forested land to nonforested land. Deforestation can be permanent, when this change is definitive, or temporary when this change is part of a cycle that includes natural or assisted regeneration.

Degraded land: land in a state that results from persistent decline or loss of biodiversity and ecosystem functions and services that cannot fully recover unaided.

Degrowth: started as an activist movement around 2008 and turned into an academic discipline, it starts from the premise that economic growth cannot be sustained *ad infinitum* on a resource constraint planet.

It demands a deep societal change, denying the need for economic growth. It is unclear "whether degrowth should be considered as a collectively consented choice or an environmentally-imposed inevitability.

Direct driver: see "driver".

Downscaling : the transformation of information from coarser to finer spatial scales through statistical modelling or spatially nested linkage of structural models.

Driver: in the context of IPBES, drivers of change are all the factors that, directly or indirectly, cause changes in nature, anthropogenic assets, nature's contributions to people and a good quality of life.

Direct drivers of change can be both natural and anthropogenic. Direct drivers have direct physical (mechanical, chemical, noise, light etc.) and behaviour-affecting impacts on nature. They include, inter alia, climate change, pollution, different types of land use change, invasive alien species and zoonoses, and exploitation.

Indirect drivers are drivers that operate diffusely by altering and influencing direct drivers, as well as other indirect drivers. They do not impact nature directly. Rather, they do it by affecting the level, direction or rate of direct drivers.

Interactions between indirect and direct drivers create different chains of relationship, attribution, and impacts, which may vary according to type, intensity, duration, and distance. These relationships can also lead to different types of spill-over effects. Global indirect drivers include economic, demographic, governance, technological and cultural ones. Special attention is given, among indirect drivers, to the role of institutions (both formal and informal) and impacts of the patterns of production, supply and consumption on nature, nature's contributions to people and good quality of life."

Drylands: drylands comprise arid, semi-arid and dry sub-humid areas. The term excludes hyper-arid areas, also known as deserts. Drylands are characterised by water scarcity and cover approximately 40 % of the world's terrestrial surface.

E

Eco-certification: programmes designed to accredit goods and services that meet defined process standards designed to improve environmental performance and, in some cases, also to improve social welfare in places of production.

Ecological (or socio-ecological) breakpoint or threshold: the point at which a relatively small change in external conditions causes a rapid change in an ecosystem. When an ecological threshold has been passed, the ecosystem may no longer be able to return to its state by means of its inherent resilience.

Ecological footprint: a measure of the amount of biologically productive land and water required to support the demands of a population or productive activity. Ecological footprints can be calculated at any scale: for an activity, a person, a community, a city, a region, a nation or humanity as a whole.

Ecological infrastructure: ecological infrastructure refers to the natural or semi-natural structural elements of ecosystems and landscapes that are important in delivering ecosystem services. It is similar to 'green infrastructure', a term sometimes applied in a more urban context. The ecological infrastructure needed to support pollinators and improve pollination services includes patches of semi-natural habitats, including hedgerows, grassland and forest, distributed throughout productive agricultural landscapes, providing nesting and floral resources. Larger areas of natural habitat are also ecological infrastructure, although these do not directly support agricultural pollination in areas more than a few kilometers away from pollinator-dependent crops.

Eco-region: a large area of land or water that contains a geographically distinct assemblage of natural communities that:

(a) Share a large majority of their species and ecological dynamics;

(b) Share similar environmental conditions, and;

(c) Interact ecologically in ways that are critical for their long-term persistence (source: WWF). In contrast to biomes, an ecoregion is generally geographically specific, at a much finer scale. For example, the "East African Montane Forest" eco-region of Kenya (WWF eco-region classification) is a geographically specific and coherent example of the globally occurring "tropical and subtropical forest" biome."

Ecosystem: a dynamic complex of plant, animal and micro-organism communities and their non-living environment interacting as a functional unit.

Ecosystem-based adaptation to climate change: the use of biodiversity and ecosystem services as part of an overall adaptation strategy to help people to adapt to the adverse effects of climate change (CBD, 2012). It refers to actions that mix the use of biodiversity and ecosystem services policy instruments with socio-economic and development policy instruments to help people adapt to the adverse effects of climate change (Scarano, 2017).

Ecosystem-based disaster risk reduction: The concept and practice of reducing disaster risks through systematic efforts to analyze and manage the causal factors of disasters, including through reduced exposure to hazards, lessened vulnerability of people and property, wise management of land and the environment, and improved preparedness for adverse events.

Ecosystem degradation: a long-term reduction in an ecosystem's structure, functionality, or capacity to provide benefits to people.

Ecosystem function: the flow of energy and materials through the biotic and abiotic components of an ecosystem. It includes many processes such as biomass production, trophic transfer through plants and animals, nutrient cycling, water dynamics and heat transfer.

Ecosystem health: ecosystem health is a metaphor used to describe the condition of an ecosystem, by analogy with human health. Note that there is no universally accepted benchmark for a healthy ecosystem. Rather, the apparent health status of an ecosystem can vary, depending upon which metrics are employed in judging it, and which societal aspirations are driving the assessment.

Ecosystem management: an approach to maintaining or restoring the composition, structure, function, and delivery of services of natural and modified ecosystems for the goal of achieving sustainability. It is based on an adaptive, collaboratively developed vision of desired future conditions that integrates ecological, socioeconomic, and institutional perspectives, applied within a geographic framework, and defined primarily by natural ecological boundaries.

Ecosystem restoration: policies and practices that are "necessarily focused on recovery of a self-sustaining living system characteristic of past or least-disturbed landscapes.

Ecosystem services : the benefits people obtain from ecosystems. In the Millennium Ecosystem Assessment, ecosystem services can be divided into supporting, regulating, provisioning and cultural. This classification, however, is superseded in IPBES assessments by the system used under "nature's contributions to people." This is because IPBES recognises that many services fit into more than one of the four categories. For example, food is both a provisioning service and also, emphatically, a cultural service, in many cultures.

Ecotourism: sustainable travel undertaken to access sites or regions of unique natural or ecological quality, promoting their conservation, low visitor impact, and socio-economic involvement of local populations.

Endangered species: a species at risk of extinction in the wild.

Endemic species: plants and animals that exist only in one geographic region.

Endemism: the ecological state of a species being unique to a defined geographic location, such as an island, nation, country or other defined zone, or habitat type; organisms that are indigenous to a place are not endemic to it if they are also found elsewhere.

Energy security: access to clean, reliable and affordable energy services for cooking and heating, lighting, communications and productive uses

Environmental additionality: the positive effect resulting from an activity or program on environmental service flows.

Environmental Impact: A measurable change to the properties of an ecosystem by a nonnative species. The logical implications of this definition are that (1) every nonnative species has an impact simply by becoming integrated into the system, (2) such impacts may be positive or negative and vary in magnitude on a continuous scale, and (3) impacts can be compared through time and across space.

Eutrophication: nutrient enrichment of an ecosystem, generally resulting in increased primary production and reduced biodiversity. In lakes, eutrophication leads to seasonal algal blooms, reduced water clarity, and, often, periodic fish mortality as a consequence of oxygen depletion. The term is most closely associated with aquatic ecosystems but is sometimes applied more broadly.

Evolutionary distinctiveness (ED): is a measure of how isolated a species or groups of species are in a phylogenetic tree. Regions with higher ED have more isolated lineages in them.

Exclusive Economic Zone (EEZ): a concept adopted at the Third United Nations Conference on the Law of the Sea (1982), whereby a coastal State assumes jurisdiction over the exploration and exploitation of marine resources in its adjacent section of the continental shelf, taken to be a band extending 200 miles from the shore. The Exclusive Economic Zone comprises an area which extends either from the coast, or in federal systems from the seaward boundaries of the constituent states (3 to 12 nautical miles, in most cases) to 200 nautical miles (370 kilometres) off the coast. Within this area, nations claim and exercise sovereign rights and exclusive fishery management authority over all fish and all Continental Shelf fishery resources.

Exotics: See "Alien species".

Extensive grazing: extensive grazing is that in which livestock are raised on food that comes mainly from natural grasslands, shrublands, woodlands, wetlands, and deserts. It differs from intensive grazing, where the animal feed comes mainly from artificial, seeded pastures.

Externality: a positive or negative consequence (benefits or costs) of an action that affects someone other than the agent undertaking that action and for which the agent is neither compensated nor penalized through the markets.

Extinction: The evolutionary termination of a species caused by the failure to reproduce and the death of all remaining members of the species; the natural failure to adapt to environmental change.

Extractives: hydrocarbons (oil and gas) and minerals.

F

Feedback: the modification or control of a process or system by its results or effects.

Food security: the World Food Summit of 1996 defined food security as existing "when all people at all times have access to sufficient, safe, nutritious food to maintain a healthy and active life".

Forest : a minimum area of land of 0.05 - 1.0 hectares with tree crown cover (or equivalent stocking level) of more than 10–30 per cent with trees with the potential to reach a minimum height of 2–5 m at maturity in situ. A forest may consist either of closed forest formations where trees of various stories and undergrowth cover a high proportion of the ground or open forest.

Forest degradation: a reduction in the capacity of a forest to produce ecosystem services such as carbon storage and wood products as a result of anthropogenic and environmental changes.

Functional diversity: the range, values, relative abundance and distribution of functional traits in a given community or ecosystem.

Functional traits: any feature of an organism, expressed in the phenotype and measurable at the individual level, which has demonstrable links to the organism's function (Lavorel et al. 1997; Violle et al. 2007). As such, a functional trait determines the organism's response to external abiotic or biotic factors (Response trait), and/or its effects on ecosystem properties or benefits or detriments derived from such properties (Effect trait). In plants, functional traits include morphological, ecophysiological, biochemical and regeneration traits. In animals, these traits include e.g. body size, litter size, age of sexual maturity, nesting habitat, time of activity.

G

Generalist species: a species able to thrive in a wide variety of environmental conditions and that can make use of a variety of different resources (for example, a flower-visiting insect that lives on the floral resources provided by several to many different plants).

Good quality of life: within the context of the IPBES Conceptual Framework – the achievement of a fulfilled human life, a notion which may varies strongly across different societies and groups within societies. It is a context-dependent state of individuals and human groups, comprising aspects such as access to food, water, energy and livelihood security, and also health, good social relationships and equity, security, cultural identity, and freedom of choice and action. "Living in harmony with nature", "living-well in balance and harmony with Mother Earth" and "human well-being" are examples of different perspectives on a "Good quality of life".

Governance: the way the rules, norms and actions in a given organization are structured, sustained, and regulated.

Grassland: type of ecosystem characterized by a more or less closed herbaceous (non-woody) vegetation layer, sometimes with a shrub layer, but – in contrast to savannas – without, or with very few, trees. Different types of grasslands are found under a broad range of climatic conditions.

H

Habitat: the place or type of site where an organism or population naturally occurs. Also used to mean the environmental attributes required by a particular species or its ecological niche.

Habitat connectivity: the degree to which the landscape facilitates the movement of organisms (animals, plant reproductive structures, pollen, pollinators, spores, etc.) and other environmentally important resources (e.g., nutrients and moisture) between similar habitats. Connectivity is hampered by fragmentation (q.v.).

Habitat degradation: a general term describing the set of processes by which habitat quality is reduced. Habitat degradation may occur through natural processes (e.g. drought, heat, cold) and through human activities (forestry, agriculture, urbanization).

Habitat fragmentation: a general term describing the set of processes by which habitat loss results in the division of continuous habitats into a greater number of smaller patches of lesser total and isolated from each other by a matrix of dissimilar habitats. Habitat fragmentation may occur through natural processes (e.g., forest and grassland fires, flooding) and through human activities (forestry, agriculture, urbanization).

Hedgerow: a row of shrubs or trees that forms the boundary of an area such as a garden, field, farm, road or right-of-way.

Human appropriation of net primary production (HANPP): the aggregate impact of land use on biomass available each year in ecosystems.

I

Impact assessment: a formal, evidence-based procedure that assesses the economic, social, and environmental effects of public policy or of any human activity.

Important Bird & Biodiversity Areas: a Key Biodiversity Area identified using an internationally agreed set of criteria as being globally important for bird populations.

Indicators: a quantitative or qualitative factor or variable that provides a simple, measurable and quantifiable characteristic or attribute responding in a known and communicable way to a changing environmental condition, to a changing ecological process or function, or to a changing element of biodiversity.

Indigenous and local knowledge systems: indigenous and local knowledge systems are social and ecological knowledge practices and beliefs pertaining to the relationship of living beings, including people, with one another and with their environments. Such knowledge can provide information, methods, theory and practice for sustainable ecosystem management.

Indigenous peoples and local communities (IPLCs): ethnic groups who are descended from and identify with the original inhabitants of a given region, in contrast to groups that have settled, occupied or colonized the area more recently. IPBES does not intend to create or develop new definitions of what constitutes “indigenous peoples and local communities"

Indirect driver: see "driver".

Institutions: encompasses all formal and informal interactions among stakeholders and social structures that determine how decisions are taken and implemented, how power is exercised, and how responsibilities are distributed.

Instrumental value See "values".

Integrated assessment models: interdisciplinary models that aim to describe the complex relationships between environmental, social, and economic drivers that determine current and future state of the ecosystem and the effects of global change, in order to derive policy-relevant insights. One of the essential characteristics of integrated assessments is the simultaneous consideration of the multiple dimensions of environmental problems.

Integrated landscape management: refers to long-term collaboration among different groups of land managers and stakeholders to achieve the multiple objectives required from the landscape.

Integrated valuation: see "values".

Intervention scenarios: see "scenarios".

Intrinsic value: see "values".

Invasive alien species: species whose introduction and/or spread by human action outside their natural distribution threatens biological diversity, food security, and human health and well-being. “Alien” refers to the species’ having been introduced outside its natural distribution (“exotic”, “non-native” and “non-indigenous” are synonyms for “alien”). “Invasive” means “tending to expand into and modify ecosystems to which it has been introduced”. Thus, a species may be alien without being invasive, or, in the case of a species native to a region, it may increase and become invasive, without actually being an alien species.”

Invasive species: see "Invasive alien species".

IPBES Conceptual Framework: the Platform’s conceptual framework has been designed to build shared understanding across disciplines, knowledge systems and stakeholders of the interplay between biodiversity and ecosystem drivers, and of the role they play in building a good quality of life through nature’s contributions to people (link to CF diagram).

IUCN protected area categorb: IUCN protected area management categories classify protected areas according to their management objectives.

IUCN Red List: The IUCN Red List of Threatened Species provides taxonomic, conservation status and distribution information on taxa that have been globally evaluated using the IUCN Red List Categories and Criteria. This system is designed to determine the relative risk of extinction, and the main purpose of the IUCN Red List is to catalogue and highlight those taxa that are facing a higher risk of global extinction (i.e. as Critically Endangered, Endangered and Vulnerable). The IUCN Red List also includes information on taxa that are categorized as Extinct or Extinct in the Wild; on taxa that cannot be evaluated because of insufficient information (i.e. are Data Deficient); and on taxa that are either close to meeting the threatened thresholds or that would be threatened were it not for an ongoing taxon-specific conservation programme (i.e. are Near Threatened).

K

Key Biodiversity Area: sites contributing significantly to the global persistence of biodiversity. They represent the most important sites for biodiversity worldwide, and are identified nationally using globally standardised criteria and thresholds.

Knowledge systems: a body of propositions that are adhered to, whether formally or informally, and are routinely used to claim truth. They are organized structures and dynamic processes (a) generating and representing content, components, classes, or types of knowledge, that are (b) domain-specific or characterized by domain-relevant features as defined by the user or consumer, (c) reinforced by a set of logical relationships that connect the content of knowledge to its value (utility), (d) enhanced by a set of iterative processes that enable the evolution, revision, adaptation, and advances, and (e) subject to criteria of relevance, reliability, and quality.

L

Land degradation: refers to the many processes that drive the decline or loss in biodiversity, ecosystem functions or their benefits to people and includes the degradation of all terrestrial ecosystems.

Land sharing: a situation where low-yield farming enables biodiversity to be maintained within agricultural landscapes.

Land sparin: also called "Land separation" involves restoring or creating non-farmland habitat in agricultural landscapes at the expense of field-level agricultural production - for example, woodland, natural grassland, wetland, and meadow on arable land. This approach does not necessarily imply high-yield farming of the non restored, remaining agricultural land. (From Rey Benayas & Bullock, 2012). See also 'Conservation agriculture' in this Glossary.

Land use: the human use of a specific area for a certain purpose (such as residential; agriculture; recreation; industrial, etc.). Influenced by, but not synonymous with, land cover. Land use change refers to a change in the use or management of land by humans, which may lead to a change in land cover.

Land use change: see "Land use".

Landscape: an area of land that contains a mosaic of ecosystems, including human-dominated ecosystems.

Leakage: an environmentally damaging activity that is relocated elsewhere after being stopped locally.

Living in harmony with nature: within the context of the IPBES Conceptual Framework – a perspective on good quality of life based on the interdependence that exists among human beings, other living species and elements of nature. It implies that we should live peacefully alongside all other organisms even though we may need to exploit other organisms to some degree.

M

Mainstreaming biodiversity: mainstreaming, in the context of biodiversity, means integrating actions or policies related to biodiversity into broader development processes or policies such as those aimed at poverty reduction, or tackling climate change.

Mangrove: group of trees and shrubs that live in the coastal intertidal zone. Mangrove forests only grow at tropical and subtropical latitudes near the equator because they cannot withstand freezing temperatures.

Maximum suatinable yield (MSY): The maximum sustainable yield (MSY) for a given fish stock means the highest possible annual catch that can be sustained over time, by keeping the stock at the level producing maximum growth. The MSY refers to a hypothetical equilibrium state between the exploited population and the fishing activity.

Megadiverse countries: 17 countries that harbor 70% of the species diversity of the planet. Seven such countries are in the Americas. In alphabetical order: Brazil, Colombia, Ecuador, Mexico, Peru, USA, Venezuela.

Meta-analysis: a quantitative statistical analysis of several separate but similar experiments or studies in order to test the pooled data for statistical significance.

Millennium Ecosystem Assessment (MEA): a major assessment of the human impact on the environment published in 2005.

Mitigation: in the context of IPBES, an intervention to reduce negative or unsustainable uses of biodiversity and ecosystems.

Models: qualitative or quantitative representations of key components of a system and of relationships between these components. Benchmarking (of models) is the process of systematically comparing sets of model predictions against measured data in order to evaluate model performance. Validation (of models) typically refers to checking model outputs for consistency with observations. However, since models cannot be validated in the formal sense of the term (i.e. proven to be true), some scientists prefer to use the words ""benchmarking"" or “evaluation"".

A dynamic model is a model that describes changes through time of a specific process.

A process-based model (also known as “mechanistic model”) is a model in which relationships are described in terms of explicitly stated processes or mechanisms based on established scientific understanding, and model parameters therefore have clear ecological interpretation, defined beforehand.

Hybrid models are models that combine correlative and process-based modelling approaches.

A correlative model (also known as “statistical model”) is a model in which available empirical data are used to estimate values for parameters that do not have predefined ecological meaning, and for which processes are implicit rather than explicit.

Integrated assessment models are interdisciplinary models that aim to describe the complex relationships between environmental, social, and economic drivers that determine current and future state of the ecosystem and the effects of global change, in order to derive policy-relevant insights. One of the essential characteristics of integrated assessments is the simultaneous consideration of the multiple dimensions of environmental problems."

Monitoring: the repeated observation of a system in order to detect signs of change.

Monoculture: the agricultural practice of producing or growing a single crop, plant, or livestock species, variety, or breed in a field or farming system at a time.

Mother Earth: an expression used in a number of countries and regions to refer to the planet Earth and the entity that sustains all living things found in nature with which humans have an indivisible, interdependent physical and spiritual relationship (see "nature").

Multidisciplinary Expert Panel (MEP): the IPBES Multidiscplinary Expert Panel is a subsidiary body established by the IPBES Plenary which oversees the scientific and technical functions ofthe Platform, a key role being to select experts to carry out assessments.

N

Native species: indigenous species of animals or plants that naturally occur in a given region or ecosystem.

Nature: in the context of IPBES, refers to the natural world with an emphasis on its living components. Within the context of western science, it includes categories such as biodiversity, ecosystems (both

structure and functioning), evolution, the biosphere, humankind's shared evolutionary heritage, and biocultural diversity.
Within the context of other knowledge systems, it includes categories such as Mother Earth and systems of life, and it is often viewed as inextricably linked to humans, not as a separate entity (see "Mother Earth").

Nature's contributions to people (NCP): all the contributions, both positive and negative, of living nature (i.e. diversity of organisms, ecosystems, and their associated ecological and evolutionary processes) to the quality of life for people. Beneficial contributions from nature include such things as food provision, water purification, flood control, and artistic inspiration, whereas detrimental contributions include disease transmission and predation that damages people or their assets. Many NCP may be perceived as benefits or detriments depending on the cultural, temporal or spatial context.

Network governance: a network is an informal arrangement where two or more autonomous individuals and/or organizations come together to exchange ideas, build relationships, identify common interests, explore options on how to work together, share power, and solve problems of mutual interest. Network governance commonly emerges when people realize that they cannot solve a particular problem or issue by working independently and that the only way to achieve their interests is by actively collaborating. Network governance varies in terms of objectives, spatial scales, leadership, representation, organization, and complexity. It is designed to supplement, not replace, other forms of natural resource governance.

Nitrogen deposition: describes the input of reactive nitrogen from the atmosphere to the biosphere both as gases, dry deposition and in precipitation as wet deposition.

Non-Indigenous Species or Non-native species or Alien species: see "invasive alien species".

Nutrient cycle: a repeated pathway of a particular nutrient or element from the environment through one or more organisms and back to the environment. Examples include the carbon cycle, the nitrogen cycle and the phosphorus cycle.

O

Ocean acidification: see "acidification".

Opportunity costs: the foregone benefits of carrying out one activity in favor of another, or giving up their initial preferred land-use plan.

Organic agricultura: any system that emphasises the use of techniques such as crop rotation, compost or manure application, and biological pest control in preference to synthetic inputs. Most certified organic farming schemes prohibit all genetically modified organisms and almost all synthetic inputs. Its origins are in a holistic management system that avoids off-farm inputs, but some organic agriculture now uses relatively high levels of off-farm inputs.

Overexploitation: harvesting species from the wild at rates faster than natural populations can recover. Includes overfishing, and overgrazing.
Participatory governance A variant or subset of governance which puts emphasis on democratic engagement, in particular through deliberative practices.

Overgrazing: an excess of herbivory that leads to degradation of plant and soil resources.

P

Participatory scenario development (and planning): approaches characterised by more interactive, and inclusive, involvement of stakeholders in the formulation and evaluation of scenarios. Aimed at improving the transparency and relevance of decision making, by incorporating demands and information of each stakeholder, and negotiating outcomes between stakeholders.

Particulate matter: a mixture of solid particles and liquid droplets (dust, dirt, soot, or smoke).

Payment for Ecosystem Services (PES): voluntary transactions that generate offsite services and are established to enable service users to pay resource providers for the conditional provision of the desired ecosystem service.

Peatlands: wetlands which accumulate organic plant matter in situ because waterlogging prevents aerobic decomposition and the much slower rate of the resulting anaerobic decay is exceeded by the rate of accumulation.

Pelagic: organisms that live in the water column.

Perennia: see "annual".

Permafrost: perennially frozen ground that occurs wherever the temperature remains below 0°C for several years.

Pesticides: A pesticide is any substance used to kill, repel, or control certain forms of plant or animal life that are considered to be pests.

Phylogenetic diversity: Phylogenetic diversity (PD) describes the breadth of evolutionary history that is represented among the organisms found in a particular area. It can capture both the diversity of ecological functions that are represented, and perhaps more importantly for human well-being, the evolutionary potential of a community to respond to future stressors.

Phylogenetic endemism: is a measure of spatial restriction of phylogenetic diversity. In other words, PE is a relative measure of endemism that represents the degree to which lineages or branches of the tree of life (calculated in my) are restricted spatially.

Plankton: aquatic organisms that drift or swim weakly. Phytoplankton are the plant forms of plankton (e.g., diatoms), and are the dominant plants in the sea. Zooplankton are the animal forms of plankton.

Plenary: within the context of IPBES – the decision-making body comprising all of the members of IPBES.

Point sources: any single identifiable source of pollution from which pollutants are discharged, such as a pipe, ditch, ship or factory smokestack.

Policy instrument: set of means or mechanisms to achieve a policy goal

Policy support tools: approaches and techniques based on science and other knowledge systems that can inform, assist and enhance relevant decisions, policy making and implementation at local, national, regional and global levels to protect nature, thereby promoting nature’s benefits to people and a good quality of life.

Poverty: poverty is a state of economic deprivation. Its manifestations include hunger and malnutrition, limited access to education and other basic services. Other corollaries of poverty are social discrimination and exclusion as well as the lack of participation in decision-making.

Primary production: Primary production is the process whereby inorganic carbon is fixed in the sunlit (euphotic) zone of the upper ocean, and forms the base of the marine food pyramid.

Prior informed consent (PIC) or free prior and informed consent (FPIC): consent given before access to knowledge or genetic resources takes place, based on truthful information about the use that will be made of the resources, which is adequate for the stakeholders or rights holders giving consent to understand the implications.

Propagule pressure: The quantity, quality and frequency of propagules (such as spores, eggs, larvae, or adults) released in a given location. This term can be seen as the introduction effort, i.e. the pool of individuals introduced in a new ecosystem/area/region and the number of times it is released.

Protected área: protected area is a clearly defined geographical space, recognized, dedicated and managed, through legal or other effective means, to achieve the long-term conservation of nature with associated ecosystem services and cultural values.

R

Ramsar site(s): a Ramsar site is a wetland site designated of international importance especially as Waterfowl Habitat under the Ramsar Convention, an intergovernmental environment treaty established in 1975 by UNESCO, coming into force in 1975.

Ramsar site refers to wetland of international significance in terms of ecology, botany, zoology, limnology or hydrology. Such site meets at least one of the criteria of Identifying Wetlands of International Importance set by Ramsar Convention and is designated by appropriate national authority to be added to Ramsar list."

Rangeland: natural grasslands used for livestock grazing.

Reducing emissions from deforestation and forest degradation (REDD+): mechanism developed by Parties to the United Nations Framework Convention on Climate Change (UNFCCC). It creates a financial value for the carbon stored in forests by offering incentives for developing countries to reduce emissions from forested lands and invest in low-carbon paths to sustainable development. Developing countries would receive results-based payments for results-based actions. REDD+ goes beyond simply deforestation and forest degradation, and includes the role of conservation, sustainable management of forests and enhancement of forest carbon stocks.

Regime shift(s): substantial reorganization in system structure, functions and feedbacks that often occurs abruptly and persists over time.

Rehabilitation: rehabilitation refers to restoration activities that move a site towards a natural state baseline in a limited number of components (i.e. soil, water, and/or biodiversity), including natural regeneration, conservation agriculture, and emergent ecosystems.

Relational value: see "values".

Remediation: any action taken to rehabilitate ecosystems.

Remote sensing: Remote sensing is the process of detecting and monitoring the physical characteristics of an area by measuring its reflected and emitted radiation at a distance from the targeted area. Special cameras collect remotely sensed imagesof the Earth, which help researchers "sense" things about the Earth.

Reports: reports shall mean the main deliverables of the Platform, including assessment reports and synthesis reports, their summaries for policymakers and technical summaries, technical papers and technical guidelines.

Resilience: the level of disturbance that an ecosystem or society can undergo without crossing a threshold to a situation with different structure or outputs. Resilience depends on factors such as ecological dynamics as well as the organizational and institutional capacity to understand, manage, and respond to these dynamics.

Resolution (spatial or temporal): see “scale”.

Richness: she number of biological entities (species, genotypes, etc.) within a given sample. Sometimes used as synonym of species diversity.

Rights-based approaches: approaches that consider international human rights law as a coherent system of principles and rules in the field of development, and uses it “as a broad guide to conducting the cooperation and aid process; social participation in that process; the obligations of donor and recipient governments; the method of evaluating aid; and the accountability mechanisms that need to be established at the local and international levels.

Route of invasion: The geographic path over which a species is transported from the donor area (origin; may be defined as Last Port of Call) to the recipient area (destination or target), which may include one or more corridors.

S

Salinization: the process of increasing the salt content in soil is known as salinization. Salinization can be caused by natural processes such as mineral weathering or by the gradual withdrawal of an ocean. It can also come about through artificial processes such as irrigation.

Savanna: ecosystem characterized by a continuous layer of herbaceous plants, mostly grasses, and a discontinuous upper layer of trees that may vary in density.

Scale: the spatial, temporal, quantitative and analytical dimensions used to measure and study any phenomenon. The temporal scale is comprised of two properties: 1) temporal extent – the total length of the time period of interest for a particular study (e.g. 10 years, 50 years, or 100 years); and 2) temporal grain (or resolution) – the temporal frequency with which data are observed or projected within this total period (e.g. at 1-year, 5-year or 10-year intervals). The spatial scale is comprised of two properties: 1) spatial extent – the size of the total area of interest for a particular study (e.g. a watershed, a country, the entire planet); and 2) spatial grain (or resolution) – the size of the spatial units within this total area for which data are observed or predicted (e.g. fine-grained or coarse-grained grid cells).

Scenario: representations of possible futures for one or more components of a system, particularly for drivers of change in nature and nature’s benefits, including alternative policy or management options.

Exploratory scenarios (also known as “explorative scenarios” or “descriptive scenarios”) are scenarios that examine a range of plausible futures, based on potential trajectories of drivers – either indirect (e.g. socio-political, economic and technological factors) or direct (e.g. habitat conversion, climate change).

Target-seeking scenarios (also known as “goal-seeking scenarios” or “normative scenarios”): scenarios that start with the definition of a clear objective, or a set of objectives, specified either in terms of achievable targets, or as an objective function to be optimized, and then identify different pathways to achieving this outcome (e.g. through backcasting).

Intervention scenarios are scenarios that evaluate alternative policy or management options – either through target seeking (also known as “goal seeking” or “normative scenario analysis”) or through policy screening (also known as “ex-ante assessment”).

Policy-evaluation scenarios are scenarios, including counterfactual scenarios, used in ex-post assessments of the gap between policy objectives and actual policy results, as part of the policy-review phase of the policy cycle. Policy-screening scenarios are scenarios used in ex-ante assessments, to forecast the effects of alternative policy or management options (interventions) on environmental outcomes."

Science-policy interfase: environment-related SPIs are organizations, initiatives or projects that work at the boundary of science, policy and society to enrich decision making, shape their participants' and audiences' understandings of problems, and so produce outcomes regarding decisions and behaviours.

Stages of invasion: Refers to the three stages that a species must successfully transit by in an invasion process and become an invasive species.

Sustainability transitions: a transformation process that is multidimensional, multistakeholder, and often operates in the long-term, by which conventional systems shift to more sustainable modes of production and consumption.

Seascape(s): seascape can be defined as a spatially heterogeneous area of coastal environment (i.e. intertidal, brackish) that can be perceived as a mosaic of patches, a spatial gradient, or some other geometric patterning (Boström et al. 2011). The tropical coastal "seascape" often includes a patchwork of mangroves, seagrass beds, and coral reefs that produces a variety of natural resources and ecosystem services.

Sector: a distinct part of society, or of a nation's economy.

Semi-natural habitat(s): an ecosystem with most of its processes and biodiversity intact, though altered by human activity in strength or abundance relative to the natural state.

Socioecological system: an ecosystem, the management of this ecosystem by actors and organizations, and the rules, social norms, and conventions underlying this management.

Soil compaction: an increase in density and a decline of porosity in a soil that impedes root penetration and movements of water and gases.

Soil degradation: the diminishing capacity of the soil to provide ecosystem goods and services as desired by its stakeholders.

Soil organic matter (SOM): matter consisting of plant and/or animal organic materials, and the conversion products of those materials in soils (ISO, 2013).

Species: an interbreeding group of organisms that is reproductively isolated from all other organisms, although there are many partial exceptions to this rule in particular taxa. Operationally, the term species is a generally agreed fundamental taxonomic unit, based on morphological or genetic similarity, that once described and accepted is associated with a unique scientific name.

Species composition: the array of species in a specific sample, community, or area.

Species distribution models: species distribution models relate field observations of the presence/absence of a species to environmental predictor variables, based on statistically or theoretically derived response surfaces, for prediction and inference. The predictor variables are often climatic but can include other environmental variables.

Species richness: the number of species within a given sample, community, or area.

Stakeholders: any individuals, groups or organizations who affect, or could be affected (whether positively or negatively) by a particular issue and its associated policies, decisions and action.

Summary for policymakers (SPM): a component of any report, providing a policy-relevant but not policy prescriptive summary of that report.

Sustainability: a characteristic or state whereby the needs of the present and local population can be met without compromising the ability of future generations or populations in other locations to meet their needs.

Sustainable Development Goals (SDGs): a set of goals adopted by the United Nations in 2015 to end poverty, protect the planet, and ensure prosperity for all, as part of the 2030 Agenda for Sustainable Development.

Sustainable use (of biodiversity and its components): the use of components of biological diversity in a way and at a rate that does not lead to the long-term decline of biological diversity, thereby maintaining its potential to meet the needs and aspirations of present and future generations.

Synergies: see "trade-off".

T

Target-seeking scenarios: see "scenarios".

Taxon: a category applied to a group in a formal system of nomenclature, e.g., species, genus, family etc. (plural: taxa).

Teleconnection: Relates to the environmental interactions between climatic systems over considerable distances.

Telecoupling: refers to socioeconomic and environmental interactions over distances. It involves distant exchanges of information, energy and matter (e.g., people, goods, products, capital) at multiple spatial, temporal and organizational scales.

Teratogen: any agent that causes an abnormality following fetal exposure during pregnancy.

Territorial Use Rights in Fisheries (TURFs): give a specific harvester exclusive access to ocean áreas.

Threatened species: in the IUCN Red List terminology, a threatened species is any species listed in the Red List categories Critically Endangered, Endangered, or Vulnerable.

Tipping point: a set of conditions of an ecological or social system where further perturbation will cause rapid change and prevent the system from returning to its former state.

Trade-off: a situation where an improvement in the status of one aspect of the environment or of human well-being is necessarily associated with a decline in or loss of a different aspect. Trade-offs characterize most complex systems, and are important to consider when making decisions that aim to improve environmental and/or socio-economic outcomes. Trade-offs are distinct from synergies (the latter are also referred to as “win-win” scenarios): synergies arise when the enhancement of one desirable outcome leads to enhancement of another.

Transhumance: a Form of pastoralism or nomadism organized around the migration of livestock between mountain pastures in warm seasons and lower altitudes the rest of the year. The seasonal migration may also occur between lower and upper latitudes. A traditional farming practice based on indigenous and local knowledge”.

Trophic level: the level in the food chain in which one group of organisms serves as a source of nutrition for another group of organisms (e.g. primary producers, primary or secondary consumers, decomposers).

Turbidity: Turbidity describes the cloudiness of water caused by suspended particles such as clay and silts, chemical precipitates such as manganese and iron, and organic particles such as plant debris and organisms.

U

Uncertainty: any situation in which the current state of knowledge is such that:

(1) the order or nature of things is unknown,

(2) the consequences, extent, or magnitude of circumstances, conditions, or events is unpredictable, and

(3) credible probabilities to possible outcomes cannot be assigned.

Uncertainty can result from lack of information or from disagreement about what is known or even knowable. Uncertainty can be represented by quantitative measures (e.g., a range of values calculated by various models) or by qualitative statements (e.g., reflecting the judgment of a team of experts)."

Units of analysis: the IPBES Units of Analysis result from subdividing the Earth’s surface into units solely for the purposes of analysis. The following have been identified as IPBES units of analisys globally:

Terrestrial:

- Tropical and subtropical dry and humid forests
- Temperate and boreal forests and woodlands
- Mediterranean forests, woodlands and scrub
- Tundra and High Mountain habitats
- Tropical and subtropical savannas and grasslands
- Temperate Grasslands
- Deserts and xeric shrublands
- Wetlands – peatlands, mires, bogs
- Urban/Semi-urban
- Cultivated areas (incl. cropping, intensive livestock farming etc.)

Aquatic, including both marine and freshwater:

- Cryosphere
- Aquaculture areas
- Inland surface waters and water bodies/freshwater
- Shelf ecosystems (neritic and intertidal/littoral zone)
- Open ocean pelagic systems (euphotic zone)
- Deep-Sea
- Coastal areas intensively used for multiple purposes by humans

These IPBES terrestrial and aquatic units of analysis serve as a framework for comparison within and across assessments and represent a pragmatic solution. The IPBES terrestrial and aquatic units of analysis are not intended to be prescriptive for other purposes than those of IPBES assessments. They are likely to evolve as the work of IPBES develops.

Urbanization: Increase in the proportion of a population living in urban areas; process by which a large number of people becomes permanently concentrated in relatively small areas, forming cities.

V

Values:

- Value systems: Set of values according to which people, societies and organizations regulate their behaviour. Value systems can be identified in both individuals and social groups (Pascual et al., 2017).
- Value (as principle): A value can be a principle or core belief underpinning rules and moral judgments. Values as principles vary from one culture to another and also between individuals and groups (IPBES/4/INF/13).
- Value (as preference): A value can be the preference someone has for something or for a particular state of the world. Preference involves the act of making comparisons, either explicitly or implicitly. Preference refers to the importance attributed to one entity relative to another one (IPBES/4/INF/13).
- Value (as importance): A value can be the importance of something for itself or for others, now or in the future, close by or at a distance. This importance can be considered in three broad classes. 1. The importance that something has subjectively, and may be based on experience. 2. The importance that something has in meeting objective needs. 3. The intrinsic value of something (IPBES/4/INF/13).
- Value (as measure): A value can be a measure. In the biophysical sciences, any quantified measure can be seen as a value (IPBES/4/INF/13).
- Non-anthropocentric value: A non-anthropocentric value is a value centered on something other than human beings. These values can be non-instrumental or instrumental to non-human ends (IPBES/4/INF/13).
- Intrinsic value: This concept refers to inherent value, that is the value something has independent of any human experience or evaluation. Such a value is viewed as an inherent property of the entity and not ascribed or generated by external valuing agents (Pascual et al., 2017).
- Anthropocentric value: The value that something has for human beings and human purposes (Pascual et al., 2017).
- Instrumental value: The value attributed to something as a means to achieving a particular end (Pascual et al., 2017).
- Non-instrumental value: The value attributed to something as an end in itself, regardless of its utility for other ends.
- Relational value: The values that contribute to desirable relationships, such as those among people or societies, and between people and nature, as in “Living in harmony with nature” (IPBES/4/INF/13).
- Integrated valuation: The process of collecting, synthesizing, and communicating knowledge about the ways in which people ascribe importance and meaning of NCP to humans, to facilitate deliberation and agreement for decision making and planning (Pascual et al., 2017)."

Vector: refers to how a species is transported, that is, the physical means or agent.

W

Water security: the capacity of a population to safeguard sustainable access to adequate quantities of and acceptable quality water for sustaining livelihoods, human well-being, and socio-economic development, for ensuring protection against water-borne pollution and water-related disasters, and for preserving ecosystems in a climate of peace and political stability.

Water stress : water stress occurs in an organism when the demand for water exceeds the available amount during a certain period or when poor quality restricts its use.

Well-being: a perspective on a good life that comprises access to basic resources, freedom and choice, health and physical well-being, good social relationships, security, peace of mind and spiritual experience. Well-being is achieved when individuals and communities can act meaningfully to pursue their goals and can enjoy a good quality of life. The concept of human well-being is used in many western societies and its variants, together with living in harmony with nature, and living well in balance and harmony with Mother Earth. All these are different perspectives on a good quality of life.

Western science: (also called modern science, Western scientific knowledge or international science) is used in the context of the IPBES conceptual framework as a broad term to refer to knowledge typically generated in universities, research institutions and private firms following paradigms and methods typically associated with the 'scientific method' consolidated in Post-Renaissance Europe on the basis of wider and more ancient roots. It is typically transmitted through scientific journals and scholarly books. Some of its central tenets are observer independence, replicable findings, systematic scepticism, and transparent research methodologies with standard units and categories.

Wetlands: areas that are subject to inundation or soil saturation at a frequency and duration, such that the plant communities present are dominated by species adapted to growing in saturated soil conditions, and/or that the soils of the area are chemically and physically modified due to saturation and indicate a lack of oxygen; such areas are frequently termed peatlands, marshes, swamps, sloughs, fens, bogs, wet meadows, etc.

Worldviews: defined by the connections between networks of concepts and systems of knowledge, values, norms and beliefs. Individual person's worldviews are moulded by the community the person belongs to. Practices are embedded in worldviews and are intrinsically part of them (e.g. through rituals, institutional regimes, social organization, but also in environmental policies, in development choices, etc.). See also 'Perceptions'; 'Concepts'; 'Reality' in this Glossary.

Z

Zoonotic diseases: or zoonoses, are directly transmitted from animals to humans via various routes of transmission (e.g. air - influenza; bites and saliva - rabies)

Annex II – Acronyms

AZE	Alliance for Zero Extinction
CaCO3	Calcium carbonate
CBD	Convencion on Biological Diversity
CITES	Convention on the International Trade in Endangered Species
CO2	Carbon dioxide
DDT	Dichlorodiphenyltrichloroethane
EbA	Ecosystem-based adaptation to climate change
EcoDRR	Ecosystem-based disaster risk reduction
EEZ	Exclusive Economic Zone
FAO	Food and Agriculture Organization of the United Nations
GDP	Gross Domestic Product
GM	Genetically modified
GMO	Genetically modified organism
HDI	Human Development Index
HIV/AIDS	Human Immunodeficiency Virus Infection / Acquired Immune Deficiency Syndrome
IBA	Important Bird and Biodiversity Areas
ICCA	Indigenous and community conserved areas
ILK	Indigenous and local knowledge
IPBES	Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services
IPCC	Intergovernmental Panel on Climate Change
IUCN	International Union for Conservation of Nature
KBA	Key Biodiversity Areas
MEA	Millenium Ecosystem Assessment
MEP	Multidisciplinary Expert Panel
NCP	Nature Contributions to People
NGO	Non-governmental Organization
OECD	Organization for Economic Cooperation and Development
PES	Payment for Ecosystem Services
PPP	Purchasing Power Parity
RCP	Representative concentration pathways
REDD	Reducing Emissions from Deforestation and Forest Degradation
REDD+	Reducing Emissions from Deforestation and Forest Degradation Plus
SDG	Sustainable Development Goals
SPM	Summary for Policy Makers
UN	United Nations
UNCCD	United Nations Conventions to Combat Desertification
UNCLOS	United Nations Convention on the Law of the Sea
UNDP	United Nations Development Programme
UNEP	United Nations Environment Programme
UNESCO	United Nations Educational, Scientific and Cultural Organization
UNFCCC	United Nations Climate Convention on Climate Change
WHO	World Health Organization