DISCLAIMER

The IPBES Global Assessment on Biodiversity and Ecosystem Services is composed of 1) a Summary for Policymakers (SPM), approved by the IPBES Plenary at its 7th session in May 2019 in Paris, France (IPBES-7); and 2) a set of six Chapters, accepted by the IPBES Plenary.

This document contains the <u>draft</u> Chapter 2.2 of the IPBES Global Assessment on Biodiversity and Ecosystem Services. Governments and all observers at IPBES-7 had access to these draft chapters eight weeks prior to IPBES-7. Governments accepted the Chapters at IPBES-7 based on the understanding that revisions made to the SPM during the Plenary, as a result of the dialogue between Governments and scientists, would be reflected in the final Chapters.

IPBES typically releases its Chapters publicly only in their final form, which implies a delay of several months post Plenary. However, in light of the high interest for the Chapters, IPBES is releasing the six Chapters early (31 May 2019) <u>in a draft form</u>. Authors of the reports are currently working to reflect all the changes made to the Summary for Policymakers during the Plenary to the Chapters, and to perform final copyediting.

The final version of the Chapters will be posted later in 2019.

The designations employed and the presentation of material on the maps used in the present report do not imply the expression of any opinion whatsoever on the part of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem 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.

IPBES Global Assessment on Biodiversity and Ecosystem Services

2.2   Chapter 2.2 Status and Trends – Nature

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Executive Summary

1. Humanity is now a dominant influence on nature worldwide *(well established)* **{2.2.5, 2.2.7}, with many impacts having accelerated rapidly in the 20th century** *(well established)* **{2.2.5.2}.** Humanity has influenced nature significantly since prehistory, both positively (e.g., development of agrobiodiversity) and negatively (e.g., extinction of megafauna and flightless island birds) *(well established)* {2.2.4, 2.2.5.1}; but nature – including species, their genes and populations, communities of interacting populations, ecological and evolutionary processes, and the landscapes and ecosystems in which they live – is now declining rapidly and many facets of nature have already been much reduced *(well established)* {2.2.5}, supporting suggestions that Earth has entered the Anthropocene.

2. Much of nature has already been lost, and what remains is continuing to decline {2.2.5.2}. Indicators of the extent and structural condition of ecosystems, of the composition of ecological communities, and of species populations overwhelmingly show net declines over recent decades; most of the exceptions are themselves symptoms of damage (e.g., the biomass of prey fish has increased, but this is because humanity has harvested most of the bigger fish that prey on them; and terrestrial vegetation biomass – though still only around half its natural baseline level – has increased slightly in recent decades, mainly because elevated CO_2 slightly increases photosynthesis) *(well established)* {2.2.5.2.1, 2.2.5.2.3, 2.2.5.2.4}. Some declines have slowed (e.g., the extent of forests is reducing less quickly than in the 1990s) and some have even been reversed (e.g., area of tree cover is increasing), but others are accelerating (e.g., most species extinction risk has arisen since 1980).

3. The degree of transformation of ecosystems from natural to human-dominated varies widely across terrestrial, inland-water and marine systems, and geographically within many systems {2.2.5.2.1, 2.2.7}. Over 40% of the world's land is now agricultural or urban, with ecosystem processes deliberately redirected from natural to anthropogenic pathways. Human drivers extend so widely beyond these areas that as little as 13% of the ocean and 23% of the land is still classified as "wilderness" – and these areas tend to be remote and/or unproductive (e.g., tundra, oceanic gyres) *(well established)* **{2.2.5.2.1}.** The most accessible and hospitable biomes either have been almost totally modified by humans in most regions (e.g., Mediterranean forests and scrub, temperate forests) or show maximum levels of conversion to anthropogenic biomes or "anthromes" (e.g., conversion of most temperate grassland to cultivated land and urban areas) *(well established)* {2.2.7.7}. Although the five freshwater and marine biomes cannot be settled and physically transformed in the same way as terrestrial biomes, they too range from unaltered to highly degraded *(well established)* {2.2.5.2.1, 2.2.7}. No global data exist on the extent of aquaculture and intensively-used coastlines, but sensitive coastal and near-shore ecosystems – such as coral reefs, mangroves and saltmarshes – are already well below natural baseline levels and continuing to decline rapidly *(established but incomplete)* {2.2.5.2.1}. Such habitats provide important resources and protection for hundreds of millions of people.

4. Globally, the net rate of loss of forests that are not managed for timber or agricultural extraction has halved since the 1990s *(established but incomplete)*, **but declines continue in the tropics** *(well established)*; **and intact forest landscapes – large areas of forest or natural mosaic with no human-caused alteration or fragmentation detectable by satellites – are still being lost from both high- and low-income countries** *(established but incomplete)* **{2.2.5.2.2}.** Forests in temperate and high latitudes have been expanding through afforestation programmes or vegetation succession after land abandonment, but the often highly biodiverse tropical primary forests continue to dwindle in most regions *(well established)* {2.2.5.2.1, 2.2.7.2}. The rate of loss of intact tropical forest landscapes has increased threefold in 10 years due to industrial logging, agricultural expansion, fire and mining *(well established)* {2.2.5.2.1}. Primary boreal and temperate forests are also increasingly degraded worldwide *(well established)* {2.2.7.3}.

5. Hotspots of rare and endemic species have on average suffered more degradation of ecosystem structure and biotic integrity than other areas, despite their importance for global biodiversity *(well established)* **{2.2.5.2, 2.2.7.15}.** Across a range of taxonomic groups, 7.3% of the land is particularly rich in species that are not found elsewhere. Indicators of ecosystem structure, community composition and species populations are ~ 20% lower in these 'hotspots' of rare and endemic species and are declining much faster (median = 74% faster), than across the world as a whole *(established but incomplete)* {2.2.5.2}. In the oceans, approximately half the live coral cover on coral reefs – among the most species-rich habitats on earth – has been lost since the 1870s, with accelerating losses in recent decades due to climate change exacerbating other drivers; live coral cover has declined by an average of 4% per decade since 1990 *(established but incomplete)* {2.2.5.2.1}.

6. Human actions threaten more species with global extinction now than ever before *(well established)* **{2.2.5.2.4}: extrapolating from detailed 'bottom-up' assessments of species in the best-studied taxonomic groups suggests that around a million animal and plant species are currently threatened, and that a third of the total species extinction risk to date has arisen in the last 25 years** *(established but incomplete)* **{2.2.5.2.4}.** Land/sea use change is the most common direct driver threatening assessed species, followed by (in descending order of prevalence) direct exploitation, pollution, invasive alien species and climate change *(well established)* {2.2.6}. The rate of species extinction is already at least tens to hundreds of times higher than it has averaged over the past 10 million years, and it is set to rise sharply still further unless drivers are reduced *(well established)* {2.2.5.2.4}. Available population trend records show widespread and rapid declines in species' distributions and population sizes *(established but incomplete)* {2.2.5.2.4}; these declines can both reduce the contributions species make to people and perturb local ecosystems with often unpredictable results**.** The prevalence of extinction risk in high-diversity insect groups is a key unknown, and knowledge of population trends is still very incomplete, especially for non-vertebrate species.

7. A 'top-down' analysis of the number of species for which sufficient habitat remains suggests that as many as half a million terrestrial species of animal and plant may already be doomed to extinction because of habitat loss and deterioration that have already taken

place *(established but incomplete)* **{2.2.5.2.4}.** These 'dead species walking' come about because responses to drivers can take many years to play out *(well established)* {2.2.5.2.4}. Habitat restoration could save many of these species if done soon after the original loss or degradation of habitat. The estimate of half a million terrestrial species, including over 3,000 vertebrate and 40,000 plant species, is produced by unprecedented integration of global environmental data with distributional information for over 400,000 terrestrial species of invertebrate, vertebrate and plant; although it is broadly consistent with the 'bottom-up' estimate of a million threatened species across the terrestrial, freshwater and marine realms, it uses entirely separate data and analysis.

8. Transformation of ecosystems to increasingly intensive human use has enabled a small fraction of species to greatly expand their distribution and increase in abundance. Invasive alien species can have devastating impacts on native species and ecosystems, particularly in areas with high endemism, disrupting the flow of nature's contributions to people. Invasive alien species can have devastating impacts on native species and ecosystems disrupting the flow of NCPs, as well as economies and human health. Over 6000 plant species are known to be invasive somewhere in the world. The number of invasive alien species and the rate of introduction of new invasive alien species seems higher than ever before and with no signs of slowing *(established but incomplete)* {2.2.5.2.3}.

9. Human actions are driving widespread changes in organismal traits *(well established)* **{2.2.5.2.5} and reductions in genetic diversity** *(established but incomplete)* **{2.2.5.2.6}. Many species are evolving rapidly as they adapt to human drivers of change, including some changes – such as resistance to antibiotics and pesticides – that pose serious risks for society** *(well established)* **{2.2.5.2.5, Box 2.5}, which evolutionary-aware policy decisions and strategies can mitigate** *(established but incomplete).* Populations have lost about 1% of their genetic diversity per decade since the mid-19th century; wild populations whose habitats have been fragmented by land-use change have less genetic diversity than those elsewhere; and mammalian and amphibian genetic diversity is lower where human influence is greater *(established but incomplete)* {2.2.5.2.6}. Although the spread of agriculture led to the development of many races and varieties of farmed animals and plants, the modernization of agriculture has seen many of these go extinct: by 2016, 559 of the 6,190 domesticated breeds of mammals used for food and agriculture (over 9 per cent) had become extinct and at least 1,000 more are threatened *(established but incomplete)* {2.2.5.2.6}. Case studies have demonstrated rapid trait changes in response to all main direct drivers and some clear examples of rapid evolution – e.g., trophy-hunted bighorn sheep have evolved smaller horns – and many species show rapid evolution in cities *(well established)* {2.2.5.2.5, Box 2.5}. Evolutionary-aware strategies can help to prevent undesirable evolution (e.g., of resistance to control measures in pests and diseases) and to promote desirable evolutionary outcomes (e.g., reduced reproduction of mosquitoes that transmit malaria) *(established but incomplete)* {Box 2.5}.

10. The global loss of forests, rates of species extinction, and average losses of originally-present biodiversity from terrestrial ecological communities all transgress proposed

precautionary 'Planetary Boundaries' *(established but incomplete)* **{2.2.5.2.1, 2.2.5.2.3}.** Transgressing these boundaries may risk tipping the Earth system out of the environmentally stable state it has been in throughout the history of civilisation, though debate about both the reality and position of the boundaries continues *(inconclusive)* {2.2.5.2.1, 2.2.5.2.3}**.** The loss of forests and tree cover (reduced to 68% and 54%, respectively, of their historical baselines) exceed the proposed Planetary Boundary for land-system change (i.e., no more than a 25% reduction in forests) *(established but incomplete)* {2.2.5.2.1}, below which the biosphere's contribution to global climate regulation may become critically compromised *(unresolved)* {2.2.5.2.1}. The global rate of species extinction is already at least tens to hundreds of times higher than the average rate over the past 10 million years and is accelerating *(established but incomplete)* {2.2.5.2.4}, exceeding the proposed boundary and potentially impoverishing the biosphere's capacity to adapt to possibly abrupt environmental change *(unresolved)* {2.2.5.2.4}. On average, terrestrial ecological communities worldwide have lost at least 20% of their originally-present biodiversity *(established but incomplete)* {2.2.5.2.3}, double the proposed safe limit beyond which the short-term healthy functioning of biomes may become compromised *(inconclusive)* {2.2.5.2.3}.

11. Land-use change has had the largest relative negative impact on nature for terrestrial and freshwater ecosystems, mainly through habitat loss and degradation; whereas in marine ecosystems, direct exploitation of organisms (mainly fishing) has had the largest relative impact, followed by land/sea-use change *(well established)* **{2.2.6.2}. The multiple components of climate and atmospheric change (e.g., changing temperature, rainfall and atmospheric CO_2 levels as well as ocean acidification) are already significant drivers of change in many aspects of nature but are not usually the most important drivers at present (***well established***) {2.2.6.2}.** The relative impact attributable to each driver also varies markedly among components of nature, taxonomic groups, regions and biomes (*established but incomplete*) {2.2.6.2, 2.2.7}. For instance, species abundance is mostly affected by land-use change in the terrestrial and freshwater systems but by direct exploitation in the marine realm. Invasive alien species often have a strong impact on oceanic island assemblages worldwide (*well established*) {2.2.3.4.1, 2.2.5.2.3}, and invasive pathogens are implicated in the rapid declines of many amphibian species (*well established*) {2.2.5.2.3}. Coral reef bleaching is a direct consequence of ocean temperature increase *(well established)* {2.2.7.15}. Temperature increase is the main factor at high latitudes both on land and in the oceans {2.2.5.2.5, 2.2.7.3, 2.2.7.5, 2.2.7.12, 2.2.7.15}. The drivers of change are all interconnected; as such they are compromising the Earth's living systems as a whole to a degree unprecedented in human history.

12. The world's major ecosystems vary in both the intensity of drivers they face and their ability to withstand them, with some close to potential collapse. The bleaching of shallow coral reefs during hotter and more frequent marine heat waves, coupled with intensifying fishing and intensification of coastline use, indicate a type of ecosystem whose thresholds of resilience are being exceeded (*well established*) {2.2.7.15}. In the Mediterranean forests, woodlands and scrub of many regions, wildfires are starting earlier in the year and increasing in number, coverage and severity which, coupled with their increasing human population due to

attractiveness for settlement and the associated expansion of urban and cultivated areas, may indicate a transformation at the biome scale *(established but incomplete)* {2.2.7.4}.

13. Many practices of Indigenous Peoples and Local Communities conserve and sustainably manage, wild and domesticated biodiversity *(well established)* **{2.2.4}. A high proportion of the world's terrestrial biodiversity lives in areas managed and/or held by Indigenous Peoples** *(well established)* **{2.2.4}, where ecosystems and ecological communities tend to be more intact and declining less rapidly than elsewhere** *(established but incomplete)* **{2.2.5.3.1}.** Practices that contribute to biodiversity include co-production of highly diverse cultural landscapes that are very heterogeneous ecologically and often rich in both wild and domesticated species {2.2.4.1, 2.2.4.2, 2.2.4.3}; contributing to agrobiodiversity by selection, domestication and maintenance of wild races and varieties of plants and animals {2.2.4.4}; traditional management practices that enhance natural resilience (e.g., by targeted burning) {2.2.4.5}; increasing landscape-scale net primary biomass production (e.g., by adaptive grazing and burning regimes) {2.2.4.6}; and protecting areas from external exploiters, e.g., slowing the spread of intensive monocrop agriculture in recognized Indigenous territories {2.2.4.7}. However, unsustainable practices are becoming increasingly common in some regions traditionally managed by these peoples and communities as lifestyles, values and external pressures change with globalization *(well established)* {2.2.4}. At least a quarter of the global land area is traditionally owned, managed[1], used or occupied by indigenous peoples. These areas include approximately 35 per cent of the area that is formally protected, and approximately 35 per cent of all remaining terrestrial areas with very low human intervention *(established but incomplete)* {2.2.5.3.1}; all these figures would rise if other local communities were considered. For the global indicators that could be compared between these Indigenous lands and the world as a whole, nature has declined by 30% less, and has declined 30% more slowly in recent years, in the Indigenous lands *(established but incomplete)* {2.2.5.3.1}.

14. Indigenous Peoples and Local Communities report that the nature important to them is mostly declining: among the local indicators developed and used by indigenous peoples and local communities, 72 per cent show negative trends in nature that underpin local livelihoods and well-being *(well established)* **{2.2.5.3.2}, which they mainly attribute to land-use change and climate change; the relative importance of these drivers varies among regions and major ecosystem types** *(established but incomplete)* **{2.2.6.3}.** Natural resource availability is generally decreasing; time needed or distance travelled to harvest resources is increasing; culturally salient species often have negative population trends; native newcomer species arrive as climate changes (e.g., southern species to arctic areas); new pests and invasive alien species colonize; natural habitats are lost, especially forests and grazing lands, while remnant ecosystems degrade and their productivity decreases; and the health condition and body size of wild animals decrease *(established but incomplete)* {2.2.5.3.2}. The drivers to which

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[1] These data sources define land management here as the process of determining the use, development and care of land resources in a manner that fulfils material and non-material cultural needs, including livelihood activities such as hunting, fishing, gathering, resource harvesting, pastoralism and small-scale agriculture and horticulture.

IPLCs most often attribute the mostly negative trends in nature (in decreasing order of prevalence and based on >300 indicators) were land-use change (e.g., tropical forest-monocrop conversions, expansion of settlements and discontinued traditional land-management practices); climatic changes, such as droughts and the increasingly unpredictable annual distribution of rainfall; arrival of new native and alien species; changing range of wild species; floods (as a combined effect of climate and land-use changes); and finally overexploitation of resources by outsiders and locals (e.g., logging and overgrazing) *(established but incomplete)* {2.2.6.3}.

15. Whereas scientific observations on the status of nature have for centuries been valued, systematically recorded, retained and synthesized in scientific outputs, Indigenous and Local Knowledge of nature has been largely disregarded, is still being lost, and has rarely been synthesised (*well established)* {2.2.2.2}. The synthesis of trends in nature observed by Indigenous Peoples and Local Communities has been hindered by the lack of regional and global institutions that would gather, aggregate and synthesize local data into regional and global summaries *(well established)* {2.2.2.2, Box 2.6}, but such efforts are emerging. Many of the aspects of nature monitored by Indigenous Peoples and Local Communities are reasonably compatible with indicators used by natural scientists but tend to be more local in scale and more directly connected to elements of nature that underpin nature's contributions to people (*well established)* {Box 2.6}, highlighting the importance of recording and synthesising them. The spread of modern lifestyles and technologies into many Indigenous and other local communities may threaten the current diversity of conceptualizations of nature and of ways of learning about and from it, as well as resource management practices that could ensure sustainable human-nature relations (*well established*) {2.2.2; 2.2.4}.

16. This global assessment has been able to make use of much more, better, more comprehensive and more representative information than was available even a decade ago *(well established)* {2.2.1}. Though uncertainties and gaps in knowledge remain, there can be no doubt that nature is continuing to decline globally *(well established)* {2.2.5, 2.2.7} in response to direct human-caused drivers *(well-established)* {2.2.6}. Some of the most important knowledge gaps are: global syntheses of Indigenous Local Knowledge about the status and trends in nature; quantitative syntheses of the status and trends of parasites, insects, microorganisms, and biodiversity in soil, benthic and freshwater environments, and of the implications for ecosystem functions; quantitative syntheses of human effects on ecosystem processes involving interactions among species, e.g., pollination; quantitative global overviews of many vital ecosystem functions; syntheses of how human impacts affect organismal traits and genetic composition; and a more comprehensive understanding of how human-caused changes to one Essential Biodiversity Variable class (e.g., ecosystem structure) ramify through to the others (e.g., community composition) and to nature's contributions to people.

2.2.1 Introduction

The definition of 'Nature' used in this assessment encompasses all the living components of the natural world. Within the context of western science, it includes biodiversity, ecosystems (both structure and functioning), evolution, the biosphere, humankind's shared evolutionary heritage, and biocultural diversity (Diaz et al. 2015). Within the context of other knowledge systems, such as those of Indigenous Peoples and Local Communities (IPLCs), Nature includes categories such as Mother Earth and systems of life, and it is often viewed as inextricably linked to humans, rather than as a separate entity (Diaz et al. 2015). IPBES's mandate includes bringing together evidence from diverse knowledge systems, including indigenous and local knowledge, and respecting diverse worldviews. Section 2.2.2 explores the diversity of worldviews and of ways in which Nature is conceptualised and outlines how they are changing.

Nature shows enormous geographic variation, at both large and small spatial scales. Associated with the range of spatial scales, there are also a broad array of institutions and governance of nature, varying from local communities through to international (Figure 2.1), which all mediate both how nature contributes to people and how people affect the state of nature (Duraiappah et al., 2014; Brondizio et al., 2009; Chapter 2.3; Chapter 2.1). At the broadest geographic scale, nature can be described according to different units of analysis (defined in Chapter 1) – from coniferous and temperate forests to tropical and sub-tropical savannas to coastal areas and deep oceans. However, within each of these units, there is variation among regions, landscapes and habitats (both terrestrial and marine) and at all levels of diversity. Section 2.2.3 tackles this complexity, organising nature's many dimensions into six classes – ecosystem structure, ecosystem function, community composition, species populations, organismal traits and genetic composition (Pereira et al. 2013) – and outlines how the global patterns of each today still largely reflects the action of natural evolutionary and ecological processes through earth's history (Whittaker et al., 2001; Willig et al., 2003; Ricklefs, 2004; Rex and Etter, 2010; Bowen et al., 2013; Pinheiro et al., 2017). Illustrative examples mostly highlight aspects of nature that underpin some of its most critical material, non-material and regulating contributions to people.

Humanity has been reshaping patterns in nature for many millennia (Lyons et al 2016;). Many IPLCs view themselves as partners in a reciprocal process of nurturing and co-production, rather than as extrinsic drivers of change (see Chapter 1). Section 2.2.4 describes the land- and sea-management practices and processes through which IPLCs have co-produced and maintained nature and continue to do so over much of the world. At least a quarter of the global land area is traditionally owned, managed[2], used or occupied by indigenous peoples (up to 60-80% if local communities are also considered). These areas include approximately 35 per cent of the area that is formally protected, and approximately 35 per cent of all remaining terrestrial areas with very low human intervention (Garnett et al. 2018).

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[2] These data sources define land management here as the process of determining the use, development and care of land resources in a manner that fulfils material and non-material cultural needs, including livelihood activities such as hunting, fishing, gathering, resource harvesting, pastoralism and small-scale agriculture and horticulture.

Whether viewed as an extrinsic driver or an intrinsic part of nature, humanity's actions now increasingly overprint the global patterns that natural processes have produced, at all scales (Figure 2.1). Section 2.2.5 considers human-caused trends in nature alongside current status. Because many anthropogenic drivers of change have intensified greatly since the mid-20th century (Chapter 2.1, Steffen et al. 2015a), the discussion of trends focuses on changes since 1970, but also briefly describes earlier positive and negative effects. As well as many science-based indicators, this section includes the first global synthesis of local trend indicators observed by IPLCs. Section 2.2.6 synthesises which of the main direct drivers – land/sea-use change, direct exploitation, climate change, pollution and invasive alien species (see Chapter 2.1) have had the greatest relative impact on nature in recent decades as judged by analysis of global indicators and the perceptions of IPLCs of the drivers behind the local changes they observe.

This subchapter's mostly global focus is balanced by brief accounts of the status, trends and drivers of change in nature within each unit of analysis (section 2.2.7), and by also highlighting three other categories of landscape that add to global nature and nature's contributions to people (NCP) disproportionately to their geographic extent: insular systems, areas particularly rich in endemic species, and hotspots of agrobiodiversity (section 2.2.3.4). The contribution of agrobiodiversity to people is obvious; but nature contributes to people in a myriad of ways, from local-scale flows of material and non-material benefits to households and communities, to global-scale regulation of the climate (Figure 2.1); Chapter 2.3 synthesises these contributions and how the trends in nature are changing them.

Synthesising and mapping variations in the state of nature across the globe and over time has been greatly facilitated by major recent advances in remote observation of biodiversity and ecosystems, in modelling and in informatics. For example, remote-sensing technologies can now provide data on ecosystem structure and function – and increasingly on abundance and distribution of biodiversity – across wide areas, with high spatial and temporal resolution (Petorelli et al. 2015), though deriving estimates of global biodiversity change from remotely-sensed data is not yet straightforward (Rocchini et al. 2015). Recording of Indigenous and local knowledge (Lundquist and Harhash 2016) can also add relevant information over smaller scales. In addition, advances in species delimitation, identification and discovery have been facilitated by new DNA technologies (e.g., Kress et al. 2015) and this in conjunction with data aggregators and repositories, such as GBIF (www.gbif.org), OBIS (www.iobis.org) and Genbank (Benson et al. 2013), make hundreds of millions of species occurrence records and gene sequences freely available. Ever-improving metadata mean that such data - despite still providing very uneven coverage taxonomically, geographically, temporally and ecologically (Akcakaya et al. 2016, Hortal et al. 2015) – can increasingly be put to a wide range of uses. This expanded biodiversity informatics landscape is increasingly well connected (Bingham et al. 2017), facilitating the synthesis of raw observations by new analytical interfaces (e.g., Ratnasingham & Hebert 2007, Jetz et al. 2012, www.iobis.org).

A growth in multi-institution collaboration has also resulted in the expansion of networks collecting parallel data, often in many countries (e.g., Kattge et al. 2011, Anderson-Teixeira et al.

2015), while the establishment of the Biodiversity Indicators Partnership and GEO BON has helped to coordinate biodiversity observations, modelling and indicators (Mace & Baillie 2007; Scholes et al. 2008; Pereira et al. 2013). The development and widespread adoption of meta-analyses and systematic reviews – facilitated by bibliographic databases, online publishing and the growth of open data – has helped researchers to synthesise previously disparate evidence (e.g., Root et al. 2003; Gibson et al. 2011). Synthesis of Indigenous and local knowledge on status and trends of nature unfortunately still lags much behind scientific synthesis, though much progress is underway in documenting local observations of trends and aggregating these to global scale (see e.g., Forest Peoples Program et al. 2016), and co-producing knowledge from ILK and science.

These developments in observation, aggregation, collaboration, modelling and synthesis mean that this global assessment has been able to draw on much better and more integrated information than was possible even only a decade ago.

Figure 2.1 – The hierarchical scales of nature, society and governance. This figure has many parallels with the IPBES conceptual framework (see Chapter 1), but emphasises how the multiple scales of governance influence both nature's contributions to people (arrows passing through the box labelled 'Ecosystem services & other goods and services) and societal feedbacks onto nature's systems. Figure from Duraiappah et al. (2014)

2.2.2 Diverse conceptualizations of nature and pluralistic knowledge systems

Nature is conceptualised differently by people having different relationships with it, including farmers, herders, fishers, hunter-gatherers, other Indigenous and local communities, urban communities, practitioners (such as hydro- and forest engineers), natural scientists, social scientists and artists. Different conceptualizations of nature lead to different types of experiential learnings and knowledge systems. Within historical times some knowledge systems such as "scientific knowledge", have gained a universal acknowledgement, while other knowledge systems such as "Indigenous knowledge" have been less well appreciated and valued, especially in terms of the information they provide on nature both locally and at larger scales.

2.2.2.1 Indigenous Peoples' and Local Communities' conceptualizations and knowledges of nature

There are many different ways that societies consider nature. There are those which consider humans as an element of nature. In contrast, others consider humans as starkly different from nature beyond the obvious biological commonalities with, and dependence on, the rest of the living world. Here we use the term 'conceptualizations of nature' to refer to views and perspectives on nature by different societies, which establish meanings to the links between humans and elements of nature, and form principles or ontologies that guide interactions with nature (Foucault 1966, Ellen 1996, Atran et al. 2002). Anthropological studies comparing many societies across the world have classified the large diversity of situations met into general models, based on the degree of continuity or separation between nature and people. Most societies that recognise a continuity between humans and nature conceptualize elements of nature as agents with an interiority, intentions or an attractivity (e.g. plants) that facilitates interactions between humans and non-human (Descola 2005, Graham 2006, Ellen 2006). Models showing strong linkages between humans and non-humans are for instance animism and totemism (Descola 2005, Sahlins 2014) . Analogism, a widespread conception of nature widely studied and typical of some Asian societies and in Europe differentiates humans and non-humans although they share some properties from microcosms (cells) to macrocosms (planets) and are made of similar elements (wind, water, fire etc.). Within such conceptualizations humans are able to find in nature many signs that guide a large set of practices, including health, food, agriculture (e.g. Friedberg 2007, Zimmerman 2011). Naturalism – the principle that theoretically characterizes modern western societies and western science – emerged with philosophers such as Descartes and emergence of modernity - conceives natural as an external element, starkly different from humans, an object of experimentation using analytical approaches for better productivity or control (Foucault 1966).

Such principles continue to influence people's attitudes to environmental and sustainability issues today. While science is therefore supposed to be neutral, Ellen (1996), shows that scientific disciplines have their own ways of conceiving the environment that serve the interest of particular groups, whether they belong to the conservation movement, have linkages to industries, churches, political parties, academics, Indigenous People, or governments. Thus, even

science and modernity establish intricate links between nature and culture and the naturalist approach is rarely void of cultural worldviews.

The IPBES Conceptual Framework puts a strong emphasis on reflecting that different societies, and different individuals within societies, have different views on desirable relationships with nature, the material versus the spiritual domain, and the present versus the past or future (Díaz et al. 2015, 2018, see also Chapter 1, section 1.3.1).

Indigenous and local knowledge systems are the knowledge of Indigenous Peoples and Local Communities who mostly live within natural and rural environments and make a living through – and define their cultural identity upon – an intimate relationship with nature, land and sea (Warren et al. 1995, Douglas et al. 1999, Sanga and Ortalli 2003, Garnett et al 2018). Indigenous knowledge systems differ from science in many ways, viewing nature holistically i.e. as said above linking all elements of nature to people in ways that enables continuities either through considering the inner self of non-humans (animism and totemism) or through common properties (analogism), all of which are linked to the social and decision-making spheres (Descola and Palsson 1996, Ellen 2002, Motte-Florac et al. 2012, Tengö et al. 2017, see more in Chapter 1). Building upon similar overall principles linking humans to nature, local knowledge systems are locally rooted, tested and culturally transmitted (Molnár and Berkes 2018). Many of these local knowledge systems vary depending on socio-cultural and religious background and also the degree of integration in modern lifestyles, a situation also encountered among Indigenous groups. For example, European small-scale multi-generational farmers, herders and fishers, and some foresters and hydro-engineers using and managing the same natural resource for generations may have strong connections to their local nature and a deep understanding of local ecological processes and may feel themselves as part of nature (Whiteman & Cooper 2000, Kis et al. 2017, Babai et al. 2014).

2.2.2.2 Collaboration between knowledge systems, changing conceptualizations

Conceptualizations of nature and related knowledge and practices are not static. They may change considerably over time at different temporal scales. Knowledge co-production between knowledge systems, interdisciplinary cooperation and modern lifestyles may accelerate change, and may foster or threaten conceptualizations and knowledge that ensure sustainable human-nature relations and consequently status and trends in nature.

Conceptualizations of nature may change in relation to levels of collaboration between knowledge systems and/ or between scientific disciplines. Although disciplinary approaches in natural or social sciences (e.g., between functional and evolutionary ecology, sociology and economics) are often still dominant, the trends towards collaborative, inter- and transdisciplinary and participatory research with stakeholders on nature and human-nature relations are now opening new options for learning. This may help develop new concepts of interactions between nature and humans that foster social-ecological systems and resilience thinking (Berkes et al. 2000), relational thinking (Chan et al. 2016), deep ecology (Naess 1973), the revisiting of the religious linkage to nature through portraying the ideas of Saint Francis of Assissi (Francis 2015)

or the pluralistic IPBES concept of nature's contributions to people (Díaz et al. 2018). Within conservation biology, views on the relationship between people and nature have continued to change over recent decades: Nature for itself, Nature despite people, Nature for people, and People and nature (Mace 2014). Some conservation biologists integrate Indigenous and local knowledge to help develop new concepts and practical actions for better conservation (Ghimire et al. 2008, Molnár et al. 2016). In ethnobiology, a discipline dedicated to study human-nature relations, there is a shift from more academic research objectives to more practical approaches including working together with Indigenous Peoples and Local Communities to co-develop sustainable management practices (Berkes 2004, Hamilton and Hamilton 2006, Newing 2011, Barrios et al. 2012).

Box 2.1. Conceptualizations of nature – examples

Conceptualisations of nature – whether Indigenous, scientific, laic, practitioner or something else – have a fundamental impact on our behaviour, relations to nature and thus on our impact on nature. Examples in this box aim to present some contrasting conceptualizations of nature.

In Indigenous conceptualizations of nature people often argue: 'All is One', 'All is connected'. April White, a Haida Indigenous artist from British Columbia created a series of prints to help negotiations of Haida fishery management with the government. These prints feature a herring-consuming predator (e.g. a whale) inside of a herring, a way reflecting the nurturing role the fish plays for so many organisms at all levels of the ecosystem. She argues that art possesses a unique storytelling power that science can stand from benefit from, "Art has a voice where a scientist might not." (Vogl, 2017).	The romantic idyllic view of nature emphasizes purity of nature, laws of nature, and harmony. This view had a huge impact on the notion of 'balance of nature' (cf. also Carlson's Silent Spring), and the development of some wilderness-oriented protected area management philosophies (source: Károly Telepy, Rocky landscape, 1870, @KOGART)

Perspectives matter. Those who experienced this view of our Earth often argue for a shift in their perspective: "You also notice how the atmosphere looks and how fragile it looks," astronaut Scott Kelly said. "It makes you more of an environmentalist after spending so much time looking down at our planet." (https://www.1045thefox.com/newsy/watching-earth-from-space-can-change-your-outlook-on-life). (Earthrise from the moon during Apollo 8, NASA)	Precision agriculture is becoming one of the dominant views about arable areas in our modern era. It aims to provide enough food for humanity with a very high level of anthropogenic assets, dominating natural processes with high-tech. This conceptualization also changes considerably our relations to the nature we manage (source: https://www.innovationtoronto.com/2016/09/precision-agriculture/).

Global processes include different contrasting tendencies such as commodification of nature, urbanization, spread of modern lifestyles, green movements, respect for the rights of Mother Nature (such as allocating personhood status to rivers), and wider acknowledgment of local space-based knowledge systems linked to complexity of socio-ecological systems. These tendencies are likely to change human-nature relations and our conceptualizations of nature. In addition, hybridization of scientific and Indigenous and local knowledge of nature is accelerating all over the world and changing our values regarding nature.

Although Indigenous and local knowledge (ILK) is locally-based, it is increasingly being shared between holder groups through local to global networks (e.g. Forest Peoples Programme et al. 2016, ICCA Consortium: www.iccaconsortium.org) and by social media.

People living in urban settings also have diverse and changing conceptualizations of nature depending on their ethnic and family history, education, religion, and their everyday experiences with urban and non-urban nature and modern technology (Loughland et al. 2003, Coyle 2005).

Scientific observations on the state of nature from a scientific perspective have for centuries been valued, systematically recorded, retained in the accumulating scientific literature and synthesised. In contrast, much Indigenous and local knowledge has not been recorded in a systematic fashion and thus much knowledge has been lost (see more in Chapter 3 and 6). This

means that records and synthesis lag far behind natural science, so there are very few resources on the status and trends of nature as observed by Indigenous Peoples and Local Communities with global coverage (Posey 1999, Forest Peoples Programme et al. 2016). Because of this imbalance, although most of the evidence in this chapter came from the context of natural sciences, a special effort has been made to also accommodate Indigenous and local knowledge on nature.

2.2.3   Overview of Nature

2.2.3.1   Essential Biodiversity Variables

Given the complexity of unit and scale when considering nature, a global system of harmonized observations has been proposed for the study, reporting, and management of biodiversity change (Pereira et al., 2013). These have been termed 'Essential Biodiversity Variables' (EBV) (see http://geobon.org/essential-biodiversity-variables/classes/) (Figure 2..2). Below we describe what is known about the current global distribution of nature using this framework, giving examples of the current knowledge on those aspects of the variables that are particularly important in terms of NCP. We then go onto discuss the contribution of Indigenous People and Local Communities to the co-production and maintenance of nature, particularly genetic, species and ecosystem diversity. This is followed by a discussion on the status and trends in nature based on these EBVs with particular emphasis on the past 50 years - trends that have resulted in the current state of nature.

A. Ecosystem structure: Units of analysis	B. Ecosystem function: Net primary production
C: Community composition: Scaled numbers of species across selected animal and plant taxa	D: Species populations: Median geographic range size of bird species
E: Species traits: Median body mass of terrestrial mammalian species	F: Genetic composition: Genetic diversity within mammalian and amphibian species

Figure 2.2. Maps of the current distribution of key aspects of nature as measured using the key metrics described in the Essential Biodiversity Variables framework. A: **Ecosystem structure** – Extent of natural and anthropogenic units of analysis considered in this assessment. B: **Ecosystem function** – Net primary production (Zhao and Running 2010; Behrenfeld et al. 1997). C: **Community composition** – Relative numbers of species per 0.5-degree grid cell, averaged across terrestrial amphibians, reptiles, mammals (IUCN spatial data) and vascular plants (Kreft et al., 2007), freshwater species (data from Collen et al., 2014) and marine species (data from Selig et al., 2014). D: **Species populations** – Median geographic range size of bird

species (Orme et al. 2006). E: **Species traits** – median body mass of mammalian species (Santini et al. 2017). F: **Genetic composition** – Average genetic diversity within mammalian and amphibian species within each grid cell (Miraldo et al. 2016).

2.2.3.2   Ecosystem structure

At the global scale, the terrestrial realm can be demarcated according a pattern of ecosystem structure (Units of Analysis) (Fig 2.2.a) where different dominant species cause the ecosystems to differ in structural complexity (e.g., tropical rainforest vs tundra or deserts) and the natural resources they can provide to people. Sometimes referred to as 'biomes' (Olson, 2001) and (for anthropogenic units) 'anthromes' (Ellis & Ramankutty 2008), the current observed units of structural complexity across the globe occur as result of processes that span millions of years and primarily reflect a combination of water-energy dynamics, geology and tectonic activity (Willis & McElwain, 2013). Demarcation of marine biomes according to ecosystem structure is an ongoing task - new habitats are still being discovered (Costello et al. 2010; Snelgrove et al. 2016) - but here too, long-term environmental and geological processes determine structure: e.g., warm-water shallow coral reefs can grow only within a narrow environmental envelope (Kennedy et al. 2013).

An understanding of global ecosystem structure is particularly important in determination of variations in photosynthetic biomass. These variations in biomass in turn have many effects on multiple aspects of NCP, from the type and quantity of material and non-material benefits available to local people, to global regulation of climates through carbon sequestration and the water cycle (Pan et al. 2011; 2013). Total photosynthetic biomass in the ocean is less than 1 % of that on land (totals of 3 PgC for marine vs 450-650 PgC on land), and this amount is mostly regulated by nutrient availability, light availability and temperature (IPCC AR5, 2013).

2.2.3.3   Ecosystem function

This term is used to describe functions provided by the stocks of materials in an ecosystem (e.g. carbon, water, minerals, and nutrients) and the flows of energy through them. The functioning of an ecosystem is therefore reliant upon a complex array of abiotic and biotic factors and underpinned by many of the variables of nature described below. When considering global ecosystem functions that are important to people, two of the most fundamental are net primary production (NPP) and carbon sequestration.

Net primary production (NPP) represents the uptake of CO_2 by plants during photosynthesis minus the amount of CO_2 that is lost during respiration. Its importance is that it provides the main source of food for non-photosynthetic organisms in any ecosystem – including humans. NPP therefore underpins many critical aspects of nature's contribution to people (Imhoff et al. 2004). Worldwide, humanity now appropriates 24% of terrestrial NPP, with over 50% being appropriated across many of the intensively farmed regions (Haberl et al. 2007). NPP shows very large spatial variation (Figure 2.2b). Terrestrial NPP varies from < 100 gC/m2/year (in polar and desert regions) to 1500 gC/m2/year in the humid tropics (Zak et al. 2007) (see also Table 2.6A), in response to levels of sunlight, temperature, water availability, CO2, nutrient availability and

the type of vegetation (Nemani et al. 2003). In the oceans, NPP is largely determined by nutrient availability (e.g. Howarth 1988; Michael et al. 2009), varying from undetectably low in nutrient-poor gyres to 500 gC/m2/year in the coastal shelves and upwelling regions.

Carbon sequestration is another critically important global ecosystem function provided by nature. This represents the difference between CO2 uptake by photosynthesis and release by respiration, decomposition, river export and anthropogenic processes such as harvesting and biomass burning. At present about 60% of the atmospheric CO_2 emitted into the atmosphere by fossil fuel emission each year (9.4 PgC / year in 2008-2017) is sequestered by nature's carbon sink in land (3.2 PgC /year in 2008-2017) and in the oceans (2.4 PgC / year in 2008-2017) (Le Quere et al., 2018), providing a vital role in regulating the Earth's climate.

Spatial and temporal patterns in carbon sinks and sources are very heterogeneous. Forest ecosystems (e.g. tropical and boreal forests) on average are carbon sinks due to CO2 fertilization, climate change, and recovery from historical land use changes (Pan et al., 2011; Kondo et al. in 2018). Between 2000 and 2007, the global forest carbon sink is estimated to have removed 2.4 billion tonnes of carbon per year from the atmosphere (Pan et al., 2011). Much of this was stored in tropical forests (0.8 billion tonnes per year), followed by temperate forests (0.8 billion tonnes per year) and boreal forests (0.5 billion tonnes per year). Soils are also an important component of terrestrial carbon sinks. For example, 50-70% of the carbon in boreal forests is stored in the soils, particularly in roots and root-associated fungi (Clemmensen et al., 2013). Furthermore, some regions, such as tropical forests and peatlands (e.g. Baccini et al. 2017) are vulnerable to becoming large CO2 emitters when there is a change in their structure and resulting function (e.g. due to land-use change).

In the ocean, CO2 is exchanged with the atmosphere primarily by air-sea exchange based on inorganic carbon chemistry. Ocean general circulation, and marine biological processes also affects CO2 exchange with atmosphere. The CO2 in the ocean is exported effectively to the deep ocean via the biological pump. Therefore, ocean NPP is one of the most essential factors to determine ocean CO2 sequestration.

2.2.3.4   Community composition

The term ecological community is used to describe an assemblage of plants, animals and other organisms that are interacting in a unique habitat where their structure, composition and distribution are determined by environmental factors such as soil type, altitude and temperature and water availability. At a global scale there is high variation in the distribution and diversity of different communities, with changes occurring across latitudinal and altitudinal gradients in both terrestrial and ocean environments. Probably one of the most well-known global trends in community composition is the latitudinal gradient in diversity on land, with the highest number of species per unit area at the equator and the lowest at the Poles (e.g. mammals, birds, reptiles, amphibians, and vascular plants) (for a review see Willig et al., 2003). Species interactions also appear to be stronger in the tropics (Schemske et al., 2009). However, some groups show departures from this trend, for example bees and aphids (Kindleman et al., 2007).

In marine environments, many groups also show a trend of decreasing species richness from the equator to the poles (e.g. fish, tunicates, crustaceans, mollusks, brachiopods, corals, foraminiferans; and see Tittensor et al. 2010), but specific groups or habitats can substantially deviate from this trend (for a review see Willig and Presley, 2018). For example, baleen whales have their highest diversity at southern subpolar and temperate latitudes (Kaschner et al. 2011). Biodiversity at the sea-floor has a maximum at or close to continental margins in areas of high carbon flux (Menot et al., 2010; Wooley et al., 2016).

Box 2.2. Global patterns in composition of marine diatoms (algae)

Marine plankton communities, including diatoms contribute around 20% of global primary productivity and are hugely significant in biogeochemical cycles and functioning of aquatic food webs (Armbrust et al., 2009). Until recently little had been known about variations in the diversity and abundance of these communities across the global oceans. A recent global study of diatoms (Malviya et al., 2016) demonstrated that although most species were found at all sites, 10 genera accounted for more than 92% of the samples indicating the dominance of a few types in the world's oceans. Overall the highest abundance of diatoms was found in regions of high productivity (upwelling zones) and the high latitude Southern Oceans.
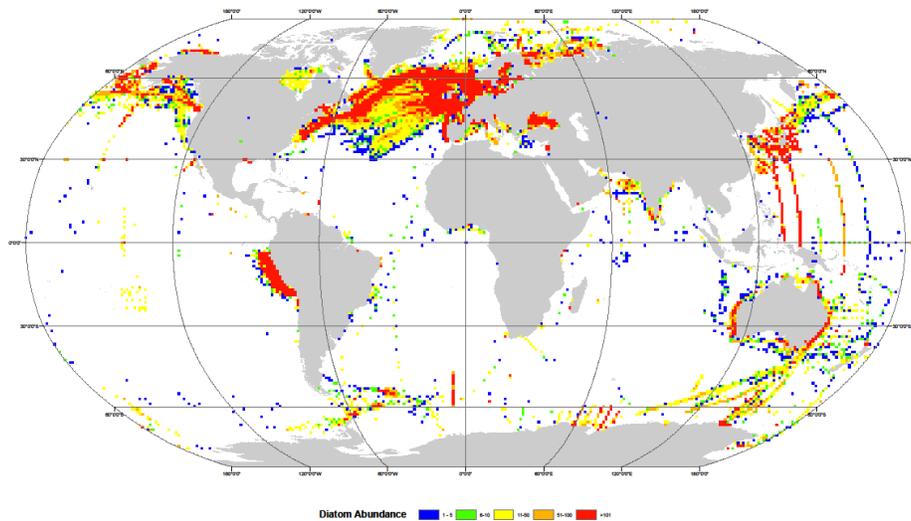
Global abundance of diatom (Bacillariophyta) species obtained from OBIS datasets [April 2018] each square is coloured according to the abundance of diatoms species observed in the area of 100 sqkm) (from Malviya et al., 2016).

In addition to these global patterns of diversity and abundance in community composition, there are also a number of well-defined communities of plants and animals associated with geographical isolation (insular systems), endemism (biodiversity hotspots), and diversity of species of plants, crops and microorganisms useful to people (agrobiodiversity hotspots). These areas are home to a disproportionately high proportion of the world's species, including for example the Eastern Arc mountains of Africa (Burgess et al., 2007) and Pacific seamounts (Richer de Forges et al. 2000); the narrow distributions of most of these species makes them intrinsically more susceptible to drivers of change. Many of these areas typically constitute only a small fraction of a biome or IPBES terrestrial and aquatic Unit of Analysis, raising the risk that their status, trends and projected futures may not be clearly reflected in assessments of nature at those large scales.

A description of each will be briefly discussed in turn.

2.2.3.4.1 Insular systems

An insular environment or "island" is any area of habitat suitable for a specific ecosystem that is surrounded by an expanse of unsuitable habitat. Examples of insular systems include mountain tops, lakes, sea-mounts, enclosed seas, and isolated islands or reefs. These systems have several important properties that set them apart from non-insular systems and thus dictate their specific consideration in this assessment.

Biotas in insular environments are often depauperate relative to biotas in similar but well-connected environments – because relatively few individuals of relatively few species arrive from across the surrounding unsuitable habitat (Vuilleumier 1970; Brown and Kodric-Brown 1977). This limited colonization results in many "empty niches" into which the few colonizing species can diversify, leading to a high proportion of endemic species (e.g. Australia, Keast 1968; Galapagos, Johnson and Raven 1973; Madagascar, Wilmé et al. 2006; mountain tops, Steinbauer et al. 2016). The result can be a collection of unique species with little or no taxonomic equivalent on the mainland, such as flightless cormorants and marine iguanas in Galapagos or honeycreepers and silverswords in Hawai'i. The limited colonization of islands can also lead to "enemy release," where the few colonists lose their defenses against former competitors, parasites, or predators, including humans. The resulting "evolutionary naïveté" renders many taxa in insular systems especially susceptible to exploitation by humans and to the spread of invasive species – especially predators and diseases (Sih et al. 2010). Examples of the resulting biological catastrophes include the whole-sale extinction of birds after the arrival of humans in New Zealand (Bunce et al. 2005, Bunce et al. 2009), the arrival of avian malaria in Hawaii (Warner 1968), and the arrival of brown tree snakes in Guam (Savidge 1987).

Many of these problems facing insular taxa are compounded when the insular habitats are very small and isolated, including tiny remote Pacific islands, alpine lakes, and dessert oases. In addition to exacerbation of these general problems of insularity, especially small insular systems often have a narrow range of environmental conditions to which local organisms are precisely adapted, along with very limited genetic variability. As a result, changing environmental conditions (e.g., climate warming or invasive alien species) that eliminate suitable habitat can be

hard to mitigate through movement or adaptive responses (e.g. Corlett and Westcott 2013; Courchamp et al. 2014; Vergés et al. 2014). Particularly obvious in this respect is the shrinking habitat of cool-climate organisms existing on mountain-top sky islands surround by unsuitable warm conditions. Finally, the small population sizes typical of species living in small insular habitats can lead to genetic drift and inbreeding that greatly reduce genetic variation in some situations. As insular taxa are often very local, rare, unique, and vulnerable, active and specific conservation efforts are critical. On the one hand, it is particularly important to limit biological invasions, as the effects for insular taxa are often severe and irreversible. On the other hand, insular taxa can often benefit from efforts to increase population sizes through habitat preservation and restoration, and to increase connectivity among isolated populations of a given species.

2.2.3.4.2 Hotspots of endemism and rarity

"Biodiversity hotspot" was a term originally proposed to describe communities of terrestrial plants and animals that contained a high concentration of endemic species yet had lost more than 70% of their original cover due to land-use change (Myers et al. 2000; Mittermeier et al., 2004; 2011). There are now 35 terrestrial hotspots that cover only 17.3% of the Earth's terrestrial surface, characterized by both exceptional biodiversity and considerable habitat loss (Marchese, 2015).

In the oceans, the concept of hotspots of endemism is less clear since a high potential for species dispersal and only a few efficient large-scale barriers hamper the development and maintenance of endemism hotspots. However, there are important exceptions from this rule and some hotspots in species richness and endemism exist. For example, the warm-water shallow coral reefs provide the habitat for estimated 8×10^5-2×10^6 species (Knowlton et al. 2010, Costello 2015) especially in the Indopacific region. They are, together with Indo-Pacific sea-mounts, vents and seeps, deep cold coral reefs, shelves around New Caledonia, New Zealand, Australia and the Southern Ocean (Ramírez-Llodra 2010, Kaiser et al. 2011), not only hotspots in species richness and functional biodiversity but also in endemism due to spatial isolation from other habitats or differences in environmental conditions. Marine range rarity is most obvious in Indo-Pacific coastal regions and off Mesoamerica (Roberts et al., 2002; Selig et al. 2014). Also, the deep-sea is rich in species and habitats (Knowlton et al. 2010), home to a conservatively estimated 5×10^5 macrofaunal species (Snelgrove and Smith, 2002).

Marine phylogenetic uniqueness is most obvious in vent and seep communities since not only single species but also larger older groups of related species (such as families) only occur in such habitats (Van Dover et al. 2018). Some of the unique macroorganisms such as the Riftia-tubeworms and vesicomyd clams depend on a symbiosis with chemosynthetic bacteria as well as archaea. Most of these marine systems need special attention because they are increasingly impacted by the exploitation of natural and mineral resources by human activities. In addition, such ecosystems are especially vulnerable due to the rarity of species in the sense of small distribution ranges and their narrow tolerance windows as a result of a strong adaptation to their environment conditions

Determining the distribution of most vulnerable species (i.e. those rare species with a small range distribution and/or ecological tolerance) is also an issue for terrestrial plants and animals. In the hotspots approach described above, which based on total richness of endemics, there tends to be an over-representation of wide-ranging species and some of the rarest and most threatened species that are range-restricted are not highlighted. It can therefore be a poor indicator of the most effective areas for targeted species conservation (Margules and Pressey 2000, Jetz and Rahbek 2002, Orme et al. 2005). An alternative approach is to use a measure such as range-size rarity (also called "endemism richness", or "weighted endemism") (Williams et al. 1996, Crisp et al. 2001, Kier and Barthlott 2001). In this approach range-size rarity is given as the count of species present in a region, weighted by their respective range proportion inside the region (Moilanen 2007, Pollock et al. 2017, Veach et al. 2017). Using this approach to determine a set of global centres of endemism richness for vascular plants, terrestrial vertebrates, freshwater fishes and select marine taxa, indicates that harmonised centres of rarity cover 7.3% of the land surface and 5% of the marine surface (Figure 2.4; for a full description of methodology and details of taxa analysed see Supplementary Material). Some of the indicators of nature reported below are sufficiently spatially resolved to allow their global status and trends to be compared to the status and trends within these
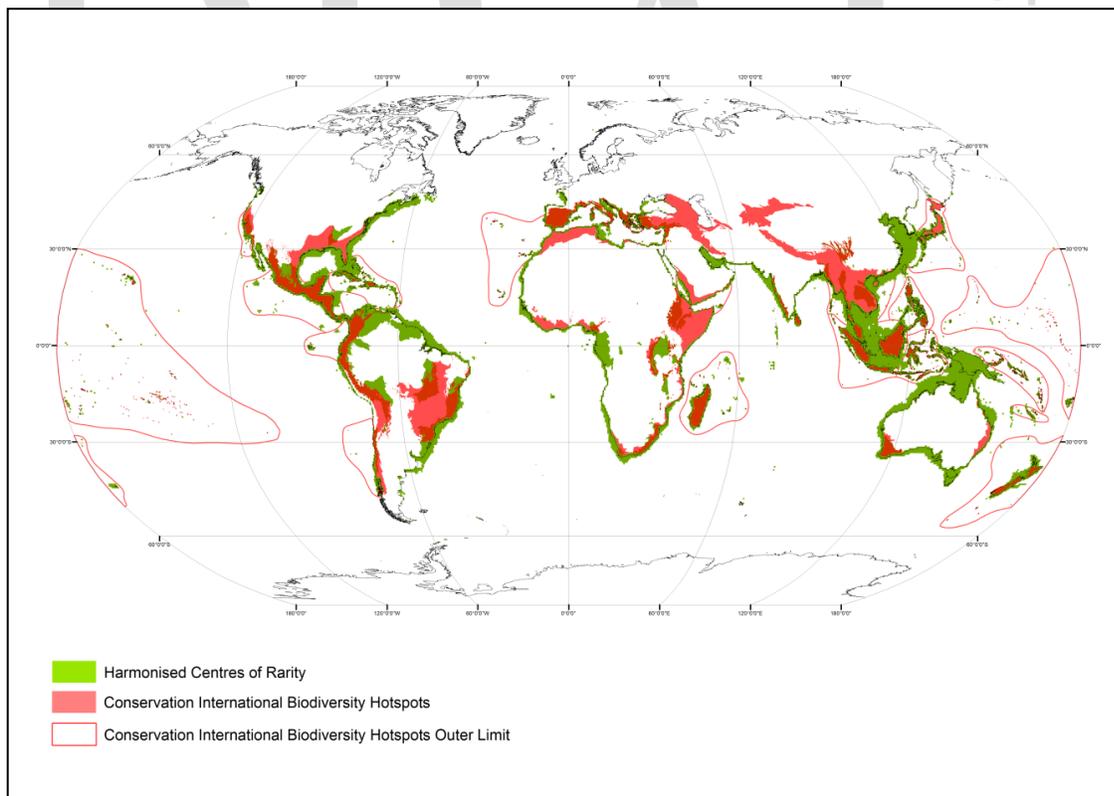

Figure 2.3: Harmonized centres of rarity, representing 7.3% of the land surface and 5% of the marine surface (see Supplementary Materials). Also indicated are the spatial extent of

Conservation International's Biodiversity Hotspots demonstrating large regions where the two measures do not overlap.

2.2.3.4.3 Hotspots of agrobiodiversity

Agrobiodiversity is the defined as "the variety and variability of animals, plants and micro-organisms that are used directly or indirectly for food and agriculture, including crops, livestock, forestry and fisheries. It comprises the diversity of genetic resources (varieties, breeds) and species used for food, fodder, fibre, fuel and pharmaceuticals. It also includes the diversity of non-harvested species that support production (soil micro-organisms, predators, pollinators), and those in the wider environment that support agro-ecosystems (agricultural, pastoral, forest and aquatic) as well as the diversity of the agro-ecosystems." (CBD, 2000). Agrobiodiversity is therefore a vital component of healthy diverse diets and of sustainable systems that provide multiple benefits to people (Biodiversity International, 2017).

Globally a very large number of crop and domestic animal species, landraces, breeds and varieties, together with their wild relatives, contribute to food security (Gepts et al. 2012; Dulloo et al., 2014; Jacobsen et al., 2015). Yet most human food comes from a relatively small number of plants and animals. Of the Earth's estimated 400,000 plant species, two-thirds of which are thought to be edible, humans only eat approximately 200 species globally (Warren, 2015), and just four crops (wheat, rice, maize and potato) account for more than 60% of global food energy intake by humans (FAO 2015a). The primary regions of diversity of major agricultural crops are mostly tropical or subtropical (Figure 2.5; Khoury et al. 2016), though many of these crops are grown well beyond their areas of origin and maximum diversity; on average, over two thirds of nations' food supplies come from such 'foreign' crops (Khoury et al. 2016). The location and conservation of hotspots of diversity of landraces, breeds and varieties therefore play a critical role in proving a gene pool and variety of traits that may provide resilience against climate change, pests and pathogens (Jacobsen et al., 2015). One branch of agrobiodiversity that has long been recognised in this respect are crop wild relatives (CWR) (Vavilov, 1926). CWRs are the ancestral species or other close evolutionary relatives from which present-day crops evolved, and they are essential to maintaining a pool of genetic variation underpinning our current crops. Their conservation is particularly important given that current crops have heavily depleted gene pools resulting from complex domestication processes, human selection and diffusions of crops and domestic animals, and ongoing diversification (Harlan, 1971, Zohary et al 2012, Vigne et al.2012, Willcox 2013, Larson & Fuller 2014, Ellis 2018, Stépanoff and Vigne 2018).

Vavilov (1926) originally recognised eight centres of crop domestication containing high numbers of CWRs. More recent mapping work (e.g. Vincent et al., 2013; Castañeda-Alvarez et al. 2016) suggests that there are many more regions where CWR occur and although the current richness hotspots align with traditionally recognised centres of crop diversity, other regions such as central and western Europe, the eastern USA, south-eastern Africa and northern Australia also contain high concentrations of richness of CWRs (Figure 2.6).

Crop wild relative taxon richness
High : 84
Low : 1
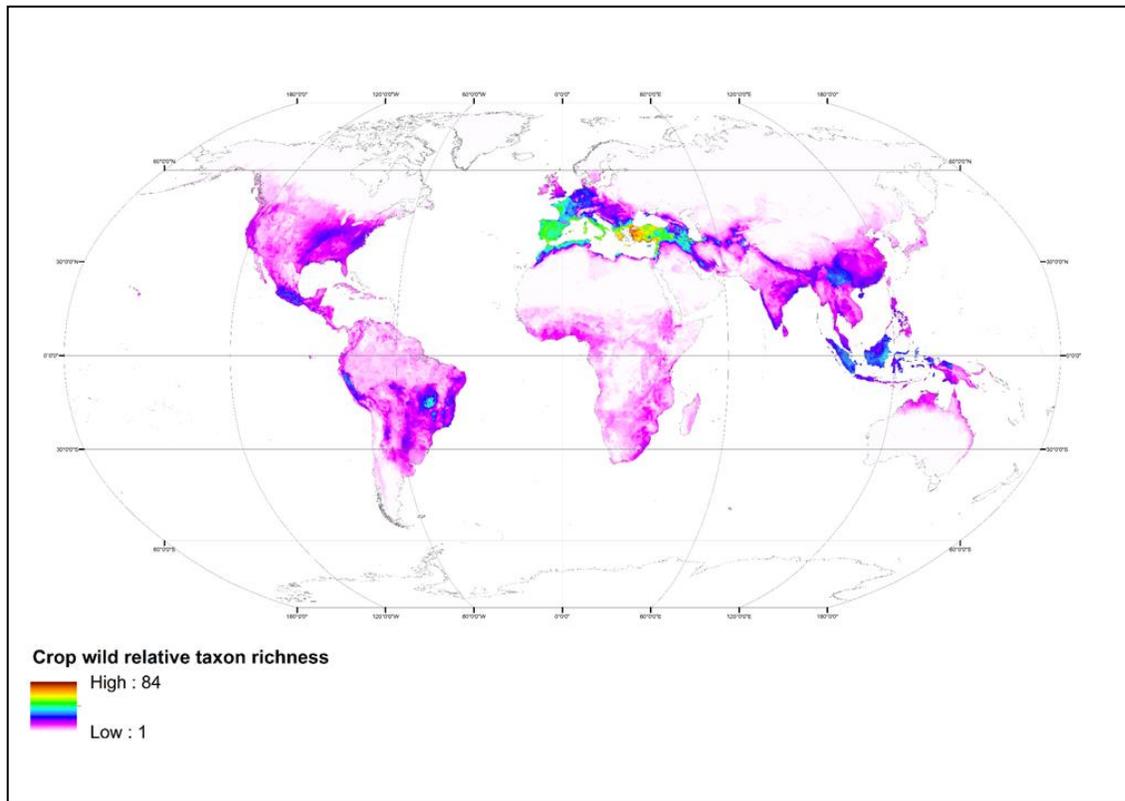
Figure 2.4 Number of crop wild relative species currently known and their global distribution (redrawn from Castañeda-Álvarez et al. 2016)

However, not all crop domestication and diversification has taken place near the areas of CWR's origins (Harlan 1971). New genomic tools and morphometric analyses are suggesting that many crops may have multi-local areas of origin (e.g. olive, wheat; (Terral and Arnold-Simard 1996, Willcox 2013) with early diffusions at a wide scale beyond the areas of origin of CWR (Figure 2.7) (see also Amazonian examples in Box 2.3). The same is also true in animal domestication, where complex evolutionary and ecological processes along with human selection have shaped the diversity and distribution of domestic animals (Larson and Fuller 2014; Larson et al., 2014) with the current distributions being much wider than original centres of origin.

Figure 2.5 Origins and primary regions of diversity of agricultural crops. Source: Khoury et al. 2016 CIAT http://rspb.royalsocietypublishing.org/content/283/1832/20160792

Another large component of agrobiodiversity underpins other material and non-material benefits (fodder, fuel, fibres etc.) (SOTWP, 2016; Diazgranados et al, 2018); for example, there are at least 28,000 plant species that are currently recorded as being of medicinal use (Alkin et al., 2017). Analysis of the distribution of these categories of plants indicates that the vast majority of them have overlapping and distinctive global ranges (see chapter 3; Figure 2.6) (Diazgranados et al., 2018; Alkin et al., 2018), yet some of the highest concentrations of medicinal plant species appear to occur in regions outside of formally designated biodiversity hotspots
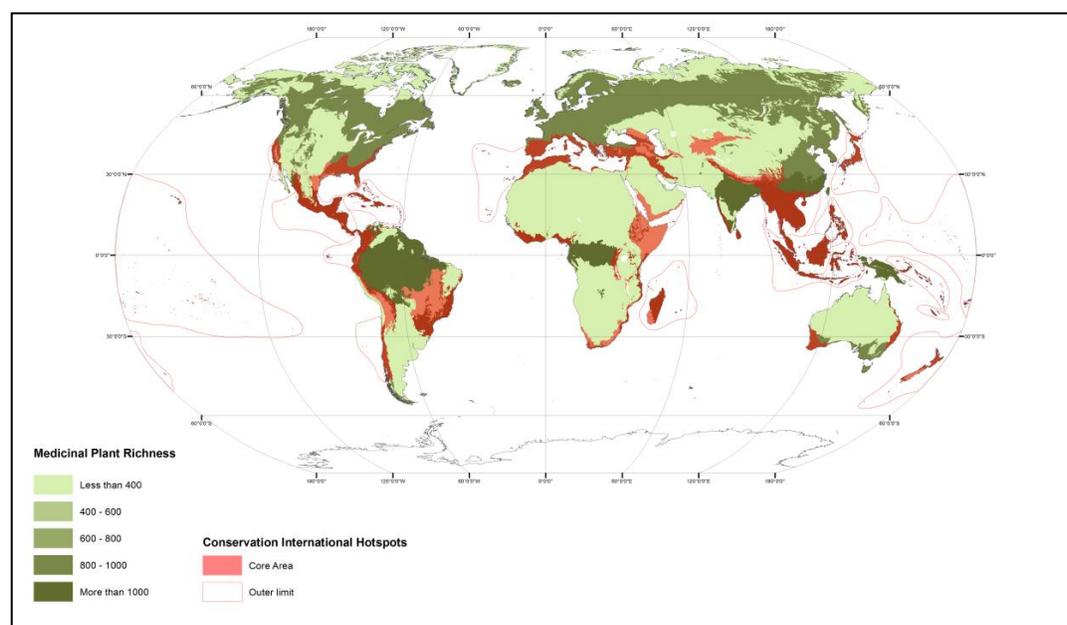
Fig 2.6 Mean medicinal plant species (per 2° grid cell) in each natural unit of analysis (Diazgranados et al., 2018; Alkin et al., 2018). Also indicated are Conservation International's Biodiversity hotspots. Acknowledgement and Source of map: Samuel Pironon et al., Department of Biodiversity Informatics and Spatial Analysis, Kew, Royal Botanic Gardens.

2.2.3.5 Species populations

A measure of the abundance and distribution of a species' population is an important facet of nature to determine because this can significantly influence the level of ecosystem service provision (Luck et al., 2003). For example, in agricultural landscapes where populations of local native vegetation provide important foraging and nesting habitats for pollinators, a distance of <2km between populations can mean that some fields are too far from nests to receive pollinator visits thus significantly reducing pollination services (Luck et al., 2003; Nogues et al., 2015). It is also an important measure to understand because species with naturally small ranges and populations tend to be more vulnerable to extinction, and the fact that a species, before going

extinct, goes through a strong reduction in population size; and because sometimes range is often used as a measure of extinction risk (see section 2.2.4).

The great majority of animal and plant species have small geographic distributions, many being found only across a very small proportion of the world's surface (e.g., Figure 2.2d; Orme et al. 2006). Species also differ in the population density (numbers per unit area or volume). This can be because of ecological and life history factors such as fecundity, trophic level and body size. For example, larger species tend to be less abundant locally, regionally and globally (White et al. 2007). Population sizes of all species can also fluctuate naturally over time and space in response to natural changes in the abiotic environment and species interactions (e.g., Inchausti and Halley 2001; Chisholm et al. 2014): as a general rule, species' abundance will tend to be higher at places and times with more resources and fewer natural enemies. This is particularly true on the deep-sea floor where abundances tend to be low even though species richness is high (Ramirez-Llodra et al. 2010).

2.2.3.6   Organismal traits

Traits refer to the structural, chemical and physiological characteristics of plants and animals (e.g., body size, clutch size, plant height, wood density, leaf size or nutrient content, rooting-depth) that are related to the uptake, use and allocation of resources. Global variations in traits reflect the combined influence of abiotic (climate, geology, soils) and biotic variables (Figure 2.2e; Simard et al. 2011) and can often mediate the relationship between organisms and their environment, thus dictating the resilience of biodiversity to environmental change (Willis et al., 2017). Many traits show consistent patterns of within-species geographic variation; for example, most mammalian and avian species show larger body size in cooler regions (Meiri and Dayan 2003; Olsen et al. 2009). Similarly, leaf area and plant height become reduced in cooler regions. An understanding of traits is important for both biodiversity conservation and determining NCP.

First, traits directly affect the ability or otherwise of plants and animals to respond to environmental perturbations including land-use change, climate change, pests and pathogens and this in turn directly affects their conservation potential. When a community of organisms faces a particular driver of change, its responses will be therefore strongly mediated by the set of traits in the community and how variation in those traits is distributed within and among species and populations (e.g., Suding et al. 2008, Diaz et al. 2013, Hevia et al. 2017). For example, in a global assessment on plant traits (Willis et al., 2017), species with a less dense wood and shorter roots were less able to withstand intervals of drought than those possessing these traits. The same is also true for animals. In a recent study on global terrestrial mammals, for example, those species not possessing traits adapted to burrowing and/or requiring a specialised diet were less resilient to climate change (Pacifici et al., 2017). There are also similar studies of traits of marine organisms to again indicate that certain traits provide greater resilience to environmental change (Costello et al. 2015).

Second, organismal traits provide a critical link to biological functions that underpin the delivery of many important societal benefits (Diaz et al., 2006; De Bello 2010; Lavorel 2013). These

include food and timber (quality and yield), pollination services, carbon sequestration, and soil nutrient quality and retention (De Bello et al., 2010). Understanding variation in traits which enable resource security and supply particularly in the face of environmental change will become increasingly important in the future (Willis et al., 2018). Yet despite their importance, still very little is known about the global distribution of traits in most taxonomic groups; e.g., a recent estimate suggested that only 2% of documented terrestrial plant species have associated trait measurements (Jetz et al., 2016).

2.2.3.7 Genetic composition

Diversity in genotypes within and between species ultimately underpins variation among plants and animals, wild and domesticated, and thus provides the essential building blocks that underpin NCP. A diverse gene pool is also critical to provide resilience to disease, climate change and other environmental perturbations both in wild and domesticated populations. Understanding the diversity and distribution of global genetic resources is therefore of critical importance and has been identified as one of the most essential biodiversity variables to monitor in order to understand the health of the planet (Steffen et al., 2017).

Factors responsible for global patterns of genetic diversity are complex and are the result of evolutionary and ecological processes occurring across multiple timescales (Schulter and Parnell, 2017). However, some generalised patterns are apparent in animals. For example, a recent study that examined genetic diversity within 4600 mammalian and amphibian species at a global scale, demonstrated a broad latitudinal gradient with higher values in the tropical Andes and Amazonia (Figure 2.2f; Miraldo et al. 2016). Other regions with high genetic diversity include the subtropical parts of South Africa for mammals and the eastern coast of Japan for amphibians. In temperate regions, western North America contains high level of genetic diversity, coinciding with high levels of mammalian species richness. In another recent study, examining genetic diversity of 76 animal species with global distributions, species traits related to parental investment and reproductive rates were also found to significantly influence genetic diversity – short-lived generalist species with high reproductive rates tend to have much higher levels of genetic diversity. Thus slow-living specialists have a much lower genetic diversity and are possibly therefore more vulnerable to environmental perturbations (Romiguier et al., 2014).

A global understanding of patterns of genetic diversity in other groups (e.g. plants, marine organisms) is largely lacking although there are many excellent regional-scale studies indicating complex patterns resulting from processes occurring over millions of years (for a review see Schulter and Parnell 2017) and gene pools associated with crop wild relatives (see above).

Policy decisions can be tailored to enhancing adaptive evolution of species that are beneficial (e.g. keystone species or species with important benefits to people) and reducing the adaptive evolution of species that are detrimental (e.g. pests, pathogens, weeds). This topic is discussed in Box 2.6 (Rapid evolution) in Section 2.2.5.2.5.

2.2.4 Contribution of Indigenous Peoples and Local Communities to the co-production and maintenance of nature

Indigenous Peoples and Local Communities (IPLCs), whose customary land encompasses approximately 50-80% of the global land area (Pearce 2016 but see problems of mapping in Chapter 1), often consider humans as an element of nature, with reciprocal exchanges between humans and non-humans that lead to nurturing and co-production.

It is important to emphasize that what has often been traditionally seen from a scientific or romantic perspective as untouched nature or wilderness is often the product of long-term use by IPLCs (e.g. the Kayapo cultural forests, Posey 1985, Willis and Birks 2006, Fairhead and Leach 1996). As wilderness areas cover an estimated 23% of land and are core to nature conservation (Watson et al. 2016), a careful re-examination of cases based on long-term paleoecological and human historical records may help to overcome this controversy.

Although global studies that compare the status of biodiversity inside versus outside IPLC areas are not yet available, a large fraction of terrestrial 'biodiversity' – perhaps up to 80% (Sobrevilla, 2008) are found on IPLC land (Gorenflo et al. 2012, Garnett et al. 2018). Whilst this figure remains an estimate until there is a more complete documentation of areas managed and/or held by IPLCs (through efforts such as the Global Registry of ICCAs) and increased inclusion of diverse governance types in the World Database on Protected Areas (Corrigan et al. 2016). However, such a high estimate is not unrealistic, given that at least a quarter of the global land area is traditionally owned, managed, used or occupied by indigenous peoples, including approximately 35 per cent of the area that is formally protected and approximately 35 per cent of all remaining terrestrial areas with very low human intervention (Garnett et al. 2018, see also Landmark.org and Chapter 1); and assuming that most rural populations pursuing small-scale non-industrial agriculture and forest management belong to 'local communities' adapted to local conditions.

It has also been noted many times that global patterns of biological diversity and cultural diversity seem not to be independent. However, while the overlap between cultural (e.g. linguistic) and biological diversity at the global scale is undeniable (Maffi 2001, Stepp et al. 2004), likely reasons for co-occurrence of linguistic and biological diversity are complex and less well known (Moore et al. 2002). Co-occurrences may be due, for example, by the longevity of local occupation, isolation caused by terrain, and specific (e.g. tribal) social structures and appear to vary among localities. Nevertheless, strong geographic concordance argues for some form of functional connection (Gorenflo et al. 2012); this is something that requires further biocultural explorations (see section 6.xx for more details) (Gavin et al. 2015).

There are many cases in the world where IPLCs 'contribute' to nature by co-producing genetic diversity, species and ecosystem diversity through 'accompanying' natural processes with anthropogenic assets (knowledge, practices, technology) (Posey 1999, Berkes 2012, Forest Peoples Programme et al. 2016). IPLCs often manage inland and coastal areas based on

culturally specific values and worldviews, applying principles and indicators like health of the land, caring for the country, and reciprocal responsibility with the goal of promoting ecosystem health, respect and integrity (Posey 1999, Berkes 2012, Lyver et al. 2017). However, unsustainable Indigenous practices are becoming increasingly common, e.g., the 'empty', 'silent' forests (cf. Redford 1991) and pasture degradation (see also 2.2.5.1-2-3 and chapter 3xx and 4xx). Changes in these areas are also often driven by changes in land management by governments and corporations (White et al. 2012), and the proportion of areas still managed by IPLCs and/or according to Indigenous and local concepts is decreasing (Borras et al. 2011). Case studies below show where the nature that contributes to people has been co-produced by local people.

2.2.4.1   Co-production of cultural landscapes with high ecosystem heterogeneity

High-diversity cultural landscapes (Agnoletti 2006) and Socio-Ecological Production Landscapes and Seascapes (SEPLS, satoyama-initiative.org), which often comprise a complex mosaic of forested areas, wet, irrigated and dry places, and coastal habitats, can provide a richness of food, fodder, timber, medicinal plants to local communities. Such landscapes have a long history of human-nature co-production. For example, the Mediterranean pasture or crop and oak agro-sylvopastoral systems (known as Dehesa in Spain, Montado in Portugal), olive and fig agro-sylvopastoral systems, holm oak-truffle woods, chestnut rural forests, and Argan agroecosystems are a number of human-nature co-production systems that are known to host a rich open habitat flora with diverse ecotones and a high level of landscape heterogeneity (Garcia-Tejero et al., 2016; Lopez-Sanchez et al., 2016, Michon 2011; Aumeeruddy-Thomas et al., 2012, 2016).

2.2.4.2   Development of species-rich semi-natural ecosystems of wild species

In cultural landscapes where people have actively changed the local disturbance regime, species-rich habitats can develop. Some of these ecosystems, made up of wild native species, became local 'hotspots' of diversity. These include for example, the European hay meadows (see Box 2.4 below) which have replaced many broad-leaved and coniferous forests in mountainous and boreal regions, and which were purposefully developed by local communities (Babai and Molnár 2014a). These meadows are among the most species-rich grasslands on Earth at several small spatial scales (up to 60-80 vascular plant species per 16 m^2, Wilson et al. 2012). The species richness of these hay meadows is correlated with the longevity and continuity of a more or less stable extensive traditional management spanning thousands of years (Zobel 1992, Merunková and Chytrý 2012; Reitalu et al. 2010).

2.2.4.3   Creation of new ecosystems with a combination of wild and domestic species

In many regions of the world Indigenous Peoples and Local Communities have combined wild and domesticated species in their agroecosystems to create new, often highly diverse ecosystems. These farming systems often sustain communities of diverse plant and animal species with increased synergy (in production and resilience). For example, IPLCs have developed multi-species tropical forest gardens in Kebu-talun and Pekarangan in West Java (Christianty et al.,

1986), rotational swidden agriculture in Thailand (Wangpakapattanawong et al. 2010 and see Box 2.4 below). In many of these locally developed traditional agroforestry systems trees, crops and/or livestock associations (Michon et al., 2000; Wiersum and Freerk, 2004) differ according to biocultural, social, economic and political contexts. In addition, the interaction between wild and cultivated components (often called rural forests) that occur in this agroforestry systems can result in hybridisation and have been suggested as a major driver of tree domestication across the planet (Aumeeruddy-Thomas, 1994; Genin et al., 2013; Michon, 2015, Aumeeruddy-Thomas & Michon 2018).

In wetland ecosystems, another combination of wild and domestic species that occurs is the rice-fish-duck culture in China (Xue et al. 2012). In addition, flooded plains across the tropics (e.g. since pre-Columbian times in Bolivia and French Guyana, also contemporary Africa) have agroecosystems based on the construction of large man-made mounds for cultivation. These are known to have brought into these flooded plains a rich agricultural biodiversity, while hosting also a large diversity of soil diversity and insects that benefit from these elevated terrestrial parts of the landscapes (McKey et al., 2016). Man-made oases or other highly modified ecosystems developed by local communities, can enhance natural processes as well as biological diversity (Tengberg et al. 2013).

Box 2.4. Two cultural landscapes where anthropogenic processes enhance biodiversity

| Embedded in the cultural landscape in Gyimes (Carpathians, Romania), these meadows were created by local Hungarian Csángó people to provide valuable hay and are now extremely species-rich semi-natural ecosystems (Section 2.2.4.2). Meadows are managed based on a deep understanding of local ecological processes (e.g., hayseed is gathered in the barns and spread onto hay meadows to increase hay quantity and | This socio-ecological production landscape has created new ecosystems with many wild and domestic species (Section 2.2.4.3), with rotational farming developed and managed by Karen people in Thailand with traditional co-creation techniques (an example for 2.2.4.3). *"A system that speaks to sustainability and livelihood security". "We select places for cultivation by listening to the sound of a stick hit to the soil in soft-wood and bamboo forests able to resprout while we avoid areas with large trees, having certain birds and mammals, and that are close to streams." "We seed not only* |

quality, Babai et al., 2014, 2015). (Photo: Dániel Babai)	*rice but many kinds of vegetables and vibrant coloured flowers believed to keep insects and birds away."* Source: Global Assessment face-to-face consultation with Kriengkrai Chechuang, Thailand.

2.2.4.4 Contributing to agrodiversity by selection and domestication

Domestication is an ongoing process that has been occurring for at least the past 20,000 years on Earth. Indigenous Peoples and Local Communities maintain many local varieties and breeds of plants, animals, and fungi and thus facilitate adaptations to the changing socio-ecological environment. Domestication is about selection of specific traits, and their integration into social-ecological niches that often differ from their original habitats. This process has occurred over millennia, since the Epipaleolithic (ca. 20 000-5 000 years ago) in the Mediterranean region and at similar periods in Papua New Guinea, Mexico, South America, and Central Asia. (Castañeda-Álvarez et al. 2016, Larson and Füller 2014, Ellis et al.2018).

Local plant and animal landraces (domesticated, locally adapted, traditional varieties and breeds) may either correspond to areas of origin or be a consequence of human-assisted dispersal across the planet. For instance, the pre-Columbian travel of sweet potato from South America where it was domesticated to the Pacific islands (Roullier et al. 2013a, b), ultimately reached Papua New Guinea where it became a very important staple food and also diversified as a result of isolation from its area of origin, new ecological conditions and selection by humans (see Box 2.3). This effect of diffusion and genetic isolation, adaptation and selection are clearly a co-production resulting from Indigenous Peoples and Local Communities manipulating ecological and biological evolutionary processes. Domestic animals have evolved far from their wild relatives' origin and represent another example of joint production linked to selection by people and adaptation to local environments. For example, there is an estimated ca. 800 local breeds of domesticated cattle, although the true numbers are incompletely known (FAO 2015a).

2.2.4.5 Enhancement of the natural resilience through traditional management

Many traditional resource management systems are 'designed' to be resilient by IPLCs, thus enabling socio-ecological systems to collectively respond or adapt to changes (Berkes and Folke 1998). Activities that are promoted to enhance natural resilience include for example, the protection and restoration of natural and modified ecosystems, the sustainable use of soil and water resources, agro-forestry, diversification of farming systems, crop development (e.g., stress-tolerant crops) and various adjustments in cultivation practices (Mijatovic et al. 2012, Barrios et al. 2012, Emperaire 2017). Farmers often utilize the diverse ecology of different crops to add synergy (such as nitrogen fixing plants, trees for shade, animals for fertilizing soils or rice fields). Such systems can diffuse risks caused by extreme climate events (e.g. floods, drought), pests or pathogens. Traditional knowledge of the ecology and cultivation of crops is combined with social practices, such as exchange networks, including seed-exchange networks (Coomes et

al. 2015, Wencelius et al. 2016, Thomas and Caillon, 2016) to increase a farmers' capacity to find adequate landraces either to adapt to changing markets or changing climate.

2.2.4.6 Increase local net primary biomass production at the landscape scale

IPLCs often increase local biomass production by, for example, rotational farming and disturbance regimes (see section 2.2.4.2 above). Examples of this type of activity includes for example, creation of rich berry patches (dominated by *Vaccinum* spp. and other berries) in boreal forests by regular burning (Johnson 1994, Davidson-Hunt 2003). In addition, prescribed regular burnings and community-based fire management of dry grasslands, forests and marshes can sometimes not only prevent larger fires that would damage local livelihoods, but they can also help the resprouting of herbaceous vegetation and restore habitat and landscape structure favourable for biodiversity (Pellatt and Gedalof 2014, Miller and Davidson-Hunt 2010, Russell-Smith et al. 2009). The same is true for some properly executed grazing regimes by domestic livestock that are adapted to the local environment and are able to prevent overgrazing (Molnár 2014, Tyler et al. 2007).

In other cases, Indigenous Peoples and Local Communities – unintentionally - maintain high levels of prey animals (e.g., sheep) that 'provide' an additional food source, which in turn are important for maintaining iconic predators (lion, leopard, wolf, bear, Casimir 2001, Mertens and Promberger 2001). Similarly, fruit gardens 'provide' food for frugivorous mammals when forest fruits are scarce (Moore et al. 2016) and thus contribute to the protection of threatened species by this extra food (Siebert and Belsky 2014).

2.2.4.7 Contribution to biodiversity by sustaining and protecting ecosystems of high conservation value from external users

IPLCs sustain naturally developed or modified ecosystems (such as the ones featured in the previous sections), and prevent species and ecosystem loss in these areas, for example by restricting access, and thus preventing unsustainable practices by outsider users (e.g. legal and illegal logging, mining, poaching, overexploitation of fisheries) (see ICCAs, OECMs, Berkes 2003, Borrini-Feyerabend et al. 2004, Corrigan et al. 2016, Nepstad et al. 2006, Govan 2016, see more in Chapter 3 and 6).

Additionally, some threatened species and some areas have strong cultural and/or spiritual significance (sacred species and sites) or are important for communities' well-being (e.g. medicinal plants, mental health) and thus have been actively conserved by communities through totem restrictions, hunting and harvesting taboos, sacred groves, rivers and springs, total or temporal use restrictions or nurturing sources of ecosystem renewal (Colding and Folke 1997, Bhagwat et al. 2012, Pungetti et al. 2012). These social taboos are often 'invisible' and thus not recognized or accounted for in conventional conservation (Colding, Folke 2001) though this is changing (Bennett et al. 2017).

2.2.5 Status and trends in nature

Nature has faced multiple drivers of change from human actions. Many of these drivers have accelerated rapidly (Chapter 2.1). The same is true for many changes in nature. Indeed, for some facets of nature, the changes have accelerated so rapidly that as much as half the total anthropogenic change in the whole of human history may have taken place since the mid-20th century. This section first discusses pre-1970 trends in nature before discussing trends since 1970 alongside current status.

2.2.5.1   Pre-1970 trends in nature

Humanity developed the capacity for significant ecosystem engineering around 10,000 years ago, marking a major ecological transition in Earth's history. Since then, the cumulative effects of human activities on some aspects of nature have been dramatic (Smith and Zeder 2013, Erlandson and Braje 2013, Boivin et al. 2016). Actions that increased the number of people the land can support have also caused species extinctions and changed species distributions, habitats and landscapes since the Stone Ages (Vitousek et al. 1997, Pimm and Raven 2000, Foley et al. 2013).

Although the state of nature has changed constantly throughout Earth history, the scale and extent of changes driven by human actions have led to this human-dominated period in Earth history being commonly called the Anthropocene (Crutzen 2002). From an ecological perspective, the Anthropocene may have begun in the late Pleistocene (Smith & Zeder 2013, Lyons et al. 2016). Human actions played a role (along with climate and other drivers) in the megafaunal extinction around the Pleistocene-Holocene boundary (Braje and Erlandson 2013, Lorenzen et al. 2014, Johnson et al. 2017); this disappearance of large herbivores and predators dramatically affected ecosystem structure, fire regimes, seed dispersal, land surface albedo and nutrient availability (Johnson 2009).

From the Late Pleistocene onwards, humans started to colonize and transformed most resource-rich landscapes on Earth (Erlandson 2013). This near-global human expansion was followed by the Neolithic spread of agriculture across the world the centres of domestication (Section 2.2.3.4.3), driven by a set of long-term, complex and independent factors like demography, climate, human behaviour and resource imbalance (Zeder and Smith 2009). This transformation to agriculture created highly modified production landscapes, caused significant land cover change (e.g., forest loss which triggered erosion and sedimentation in rivers and lakes), and spread new varieties and breeds of domesticated animals and crops as well as other (e.g. weed) species (Baker 1991). These changes altered all earth systems from the lithosphere and biosphere to the atmosphere. For example, expansion of paddy rice fields and pastoralism is thought to have increased atmospheric methane from as early as 4000 years ago (Fuller et al. 2011).

All these changes increasingly concentrated biomass into human-favoured species (Williams et al. 2015, Barnosky 2008). Humans used fire for large-scale transformation of "savannas" (Archibald et al. 2012), while diverse grazing regimes reshaped and expanded grasslands during the last 3000-7000 years. Improved seafaring since the mid-Holocene enabled colonization of

even remote islands. Island ecosystems, with "naïve" species and low functional redundancy, often changed dramatically after human colonization (Rick et al. 2013); e.g. two-third of bird species native to Pacific islands went extinct between initial human colonization (after 1300 BC) and European contact (17th century) (Duncan et al. 2013). Many exploited species worldwide have evolved to be smaller (Fitzpatrick and Keegan 2007, Jorgensen et al. 2007).

European colonialism from 1500 to early 1800s fundamentally transformed pre-existing indigenous cultural landscapes, with deforestation for monocrop plantations and the spread of invasive alien species (Dyer et al. 2017). Populations of fur animals, fishes and whales were overexploited for the new global market (Lightfoot et al. 2013, Monsarrat et al. 2016, Rodrigues et al. 2018). Spread of global commerce mostly from Europe, together with the spread of the European naturalistic worldview, had a huge impact on local human-nature relations and hence on land use (Lightfoot et al. 2013), resulting for example in the spread of timber-oriented forest management (Agnoletti 2006). Global forest cover decreased for millennia (Pongratz et al. 2008), and large trees were lost from many areas well before the mid-20th century (Lindenmayer et al. 2012, Rackham 2000).

Marine defaunation started only a few hundred years ago and may have been less severe than defaunation on land (McCauley et al. 2015, Dirzo et al. 2014). Though few marine species are known to have gone globally extinct (Webb & Mindel 2015), many became ecologically or commercially extinct with the onset of commercial and industrial scale exploitation, the most threatened animals being those that directly interact with land (McCauley et al. 2015).

The Industrial Revolution in Europe, and the growth of populations and cities that it enabled, accelerated impacts on biodiversity. For example, some habitats have lost >90% of their area since 1800 especially in Europe (Biró et al. 2018) and North America. The Green Revolution after WWII drove further agricultural intensification, causing a rapid decline of species of agricultural habitats and the spread of invasive species, and further increasing the proportion of net primary production taken by humanity (Krausmann et al. 2013). Extinction rates rose sharply in the 20th century for all taxonomic groups for which a robust assessment can be made (Figure 2.7).
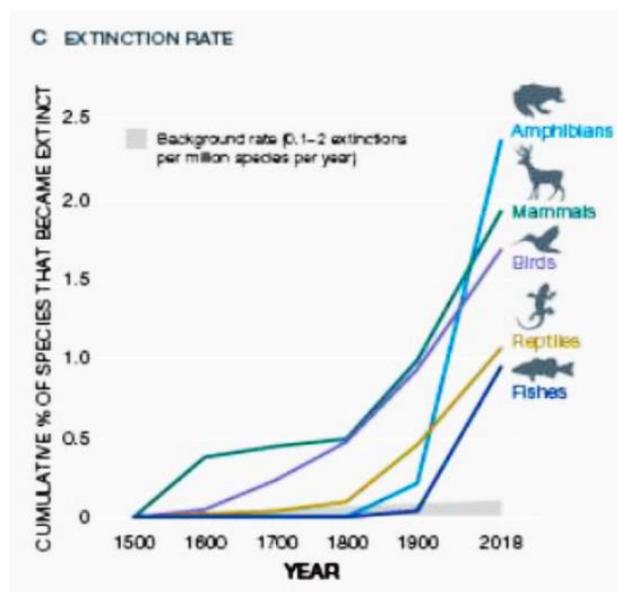
Figure 2.7. Extinction rates per century since 1500 for vertebrate classes. Fishes includes bony fishes, cartilaginous fishes and lampreys. Values for Reptiles and Fishes are likely to be underestimates as not all species in these groups have been assessed for the IUCN Red List. The range of background rates of extinction (grey line) is based on 0.1- 2 extinctions per million species per year, following Ceballos et al. (2015) and references therein. Source: Analysis of data in the IUCN Red List in September 2018.

2.2.5.2   Trends in nature since 1970 and current status

The status and recent trends seen in terrestrial, freshwater and marine ecosystems clearly show that humanity is a dominant global influence on Nature. This assessment of current status and trends since 1970 synthesises over 50 quantitative global indicators, covering an unprecedentedly diverse set of facets of nature (because nature is too complex for its trends and status to be captured by one or a few indicators: Section 2.2.3), together with recent meta-analyses, reviews and case studies, organised into Essential Biodiversity Variable classes (Section 2.2.3.1). Attribution of changes to drivers is considered in Section 2.2.6. below.

The linkages among different aspects of nature in ecosystems mean that trends may differ systematically among EBV classes. For instance, forest loss causes local extinction of forest-adapted species, but this species may accelerate once the fraction of natural habitat remaining goes below 30% (Banks-Leite et al. 2014; Ochoa-Quintero et al. 2015). Likewise, local declines in species richness can drive non-linear declines in ecosystem function, with function initially declining less rapidly than species richness (Hooper et al. 2012, Cardinale et al. 2012).

Even within an EBV class, indicator trends are likely to vary by much more than their statistical margins of error. One reason is that some components of nature are expected to be more sensitive than others – e.g., habitats such as warm-water coral reefs that have narrow environmental tolerances – so indicators reporting on them may show the steepest trends; they are in effect the 'canaries in the coal mine' that provide the first clear evidence that drivers are

reshaping nature. By contrast, other indicators try to reflect the status of nature more broadly, e.g., all species within a large taxonomic group such as mammals; these indicators are also important because the broader state of nature underpins consistent delivery of many NCP, especially over longer time scales, across larger areas, and in the face of ongoing drivers (Cardinale et al. 2012; Mace et al. 2012; Oliver et al. 2015; Steffen et al. 2015b; Winfree et al. 2018). A second reason for variation is that some indicators use more coarse-grained data than others. For example, species' extinction risk is measured on a relatively coarse spatial and temporal scale (the IUCN Red List categories), so indicators synthesising these data may miss gradual declines of abundant, widespread species, which indicators based on species' abundances may capture (Butchart et al. 2005). Consequently, indicators of species populations based on species' extinctions and extinction risk are here considered separately from those based on species' abundances or distributions. A third reason is that some trends might only be apparent at one spatial scale. Because this is particularly true for community composition (McGill et al. 2015, Jarzyna & Jetz 2018), trends within this EBV class are discussed at three different scales: local (e.g., the set of species in a small area of the same habitat type), regional (e.g., the set of species in a country or large grid cell), and the differences between local communities within the same region.

Where possible, each indicator is expressed in two ways. First, the recent rate of change shows how quickly it is changing over time; the average per-decade change in the indicator is expressed as a percentage of the estimated value for 1970 (or, if later, for the beginning of the time-series). Second, the current status is shown as a percentage of the inferred or estimated natural baseline level (i.e., the value in a pristine or at least much less impacted – e.g., pre-industrial – world), showing how much remains (see Figures 2.8-2.20). Most indicators are designed such that a larger value equates to there being more of the focal component of nature, but some are the other way round (e.g., numbers of species extinctions). Here, for ease of comparisons, such reverse indicators are rescaled so that values are larger when there is more nature. (Note that more is not always better – for instance, a rise in the number of invasive alien species is not desired.)

For some indicators that can be mapped at sufficient spatial resolution, the status and trend are also shown within the hotspots of narrowly-distributed species (mapped in Figure 2.3), and within the areas mapped (Garnett et al. 2018) as Indigenous lands (mapped in Figure SPM6); in the plots below, these have "Hotspots" or "Indigenous Lands" as part of the indicator name. Some other indicators are also subsets (e.g., the persistence of pollinating vertebrates is a subset of the persistence of all terrestrial vertebrates). All subsets are shown as unfilled symbols in the plots that follow; to avoid 'double counting', they are omitted when calculating averages across indicators. The appendix INDICATOR STATUS AND TRENDS defines and explains each indicator, its source and how it has been treated here, along with (where possible) how the natural baseline was estimated and plots of how the indicators has changed over time. In this section, italics are used to highlight indicators plotted in the figures for each Essential Biodiversity Variable class. Chapter 3 considers many of the same indicators, sometimes with very different presentation and analysis reflecting that chapter's different scope. Indicators that

are designed to report on trends in nature directly responsible for particular classes of NCP are developed and presented in Chapter 2.3.

2.2.5.2.1   Ecosystem structure

(N.B. Italics denote indicators plotted in Figure 2.8.)

Most global indicators show a net deterioration in the structure (i.e., extent and physical condition) of natural ecosystems since 1970 of at least 1% per decade (Figure 2.8A), and indicators have fallen to by almost half of their natural baseline levels (to a median of 53.2%: Figure 2.8B). There can be no doubt that human actions have radically changed, and are continuing to change, ecosystem structure – especially in sensitive ecosystems – across much of the world. Given that ecosystem structure sets the stage for ecological, evolutionary and social-ecological processes, these changes potentially jeopardise nature's ability to deliver many societal benefits. The indicators that can be estimated within the terrestrial hotspots of rare species have lower status and steeper declines there than across the globe, which is particularly concerning for biodiversity conservation; conversely, these indicators have better current status and slower declines in Indigenous lands than globally.
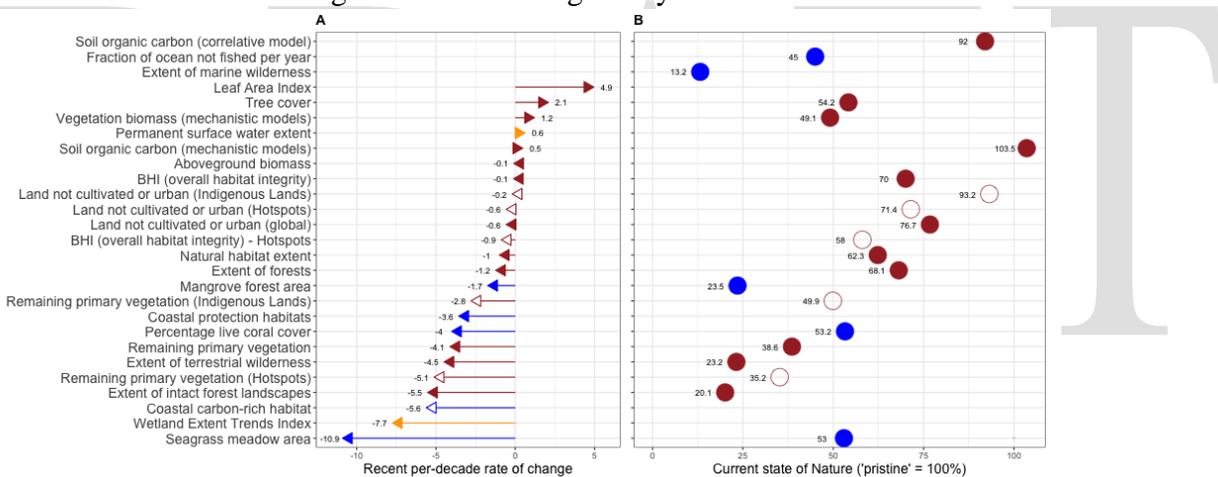
Figure 2.8: Global indicators of natural ecosystem structure. Marine indicators are in blue, terrestrial in brown and freshwater in orange. Solid symbols are used for overall indicator values, whereas hollow symbols show the indicator is a subset of another. **A**: Trends, shown as the average per-decade rate of change since 1970 (or since the earliest post-1970 year for which data are available), ordered by rate of change. Most indicators show declines (left-pointing arrows; 14/17 overall indicators) and the median change overall is -1.1% per decade. **B**: Estimated current status relative to a pristine or at largely pre-industrial baseline. On average, status is only just over half of the baseline value (median = 53.2%). Note that, even though tree cover has a positive trend in recent decades, earlier declines mean it is still well below its natural baseline. Some indicators provide only either rate or status so appear in only one panel. Supplementary Material for Chapter 2 Nature has detailed information and full references for each indicator, including subsets.

Indicators of coastal and shallow marine ecosystems are already at low levels and are continuing to decline particularly rapidly (e.g., *seagrass meadow area* (Waycott et al. 2009), *mangrove forest area* (Hamilton & Casey 2016), *live coral cover* on reefs (Ortiz et al. 2018, Eddy et al. 2018). The declines have direct societal implications. For example, *coastal protection habitats* (Ocean Health Index 2018) protect against storm surges and can elevate coastlines in step with rising sea level (Spalding et al. 2016), and *coastal carbon-rich habitats* (Ocean Health Index 2018) can act as carbon sinks.

Other sensitive ecosystems also combine rapid decline with low levels relative to historical baselines. For example, only 13% of ocean (including almost none of most coastal ecosystems) (Jones et al. 2018) and 23% of land (most of it inhospitable or remote) (Watson et al. 2016) are sufficiently free of obvious human impacts to still be classed as *wilderness* (and see 2.2.4 for discussion of likely human influence even there). *Intact forest landscapes* (defined as areas of forest or natural mosaics larger than 500km^2 where satellites can detect no human pressure) continue to decline rapidly in both rich and poor countries, and especially in the Neotropics, due to industrial logging, agricultural expansion, fire and mining (a loss of 7% between 2000 and 2013: Potapov et al. 2017). Estimates of the fraction of land that can still be viewed as 'natural' rather than anthropogenic range from under 25% (Ellis & Ramankutty 2008) to over 50% (Sayre et al. 2017; FAO 2013), depending on how 'natural' is defined. Just 39% of land area is still classed as *primary vegetation* (i.e., has never been cleared or regularly grazed) (Hurtt et al. 2018), putting many species of habitat specialists at potential risk (Brook et al. 2003, Matthews et al. 2014). The *Biodiversity Habitat Index* (Hoskins et al. 2018), which recognises that modified habitat still supports some biodiversity, estimates the current global integrity of terrestrial habitat for native biodiversity to be 70% of its original natural level. The *Wetland Extent Trend Index* is declining rapidly (Dixon et al. 2016) and as much as 87% of the natural wetland in 1700 was lost by 2000 (Davidson 2014) (see also 2.2.7.9). The slight net increase in the *extent of permanent surface water* masks extensive turnover: 13% of the area of permanent water in the 1980s had been lost by 2015, outweighed by a 16% expansion largely from new reservoirs (Pekel et al. 2016).

Although *land neither cultivated nor urban* (based on satellite data and including grazing land (ESA 2017)) has decreased only slowly since 1992, much more rapid declines are seen in some units of analysis (temperate grasslands, -2.5%; tropical and subtropical forests, -1.3%; see Supplementary Material for Chapter 2 Nature). Some regions have also seen particularly rapid land cover change: between 2001-2012, the Arctic saw a 52% increase in the extent of forest, 19% increase in wetland and a 91% decrease in barren ground (Shuchman et al. 2015).

Another indicator with marked regional variation is *aboveground biomass* (Figure 2.9): globally, it fell by only ~ 0.2% (< 1 PgC) between 1990 and 2012 (with a dip in the mid-2000s), but tropical forests saw a fall of ~ 5 PgC (especially in Amazonia and Southeast Asia) while boreal and temperate mixed forests saw a rise of ~ 2.3 PgC (Liu et al. 2015). Land-use change and intensification have reduced *vegetation biomass* – of which trees are the main component – to below 50% of the level expected if there were no human land use, mostly before 1800 (Erb et al.

2017), with model-ensemble estimates (Le Quere et al. 2018) showing an upward trend since 1970 driven by CO_2 fertilization, climate change and regrowth after previous land-use change.

Mg C ha-1 yr-1
2.0
-2.0
Mean annual change in aboveground biomass carbon between 1992 and 2012
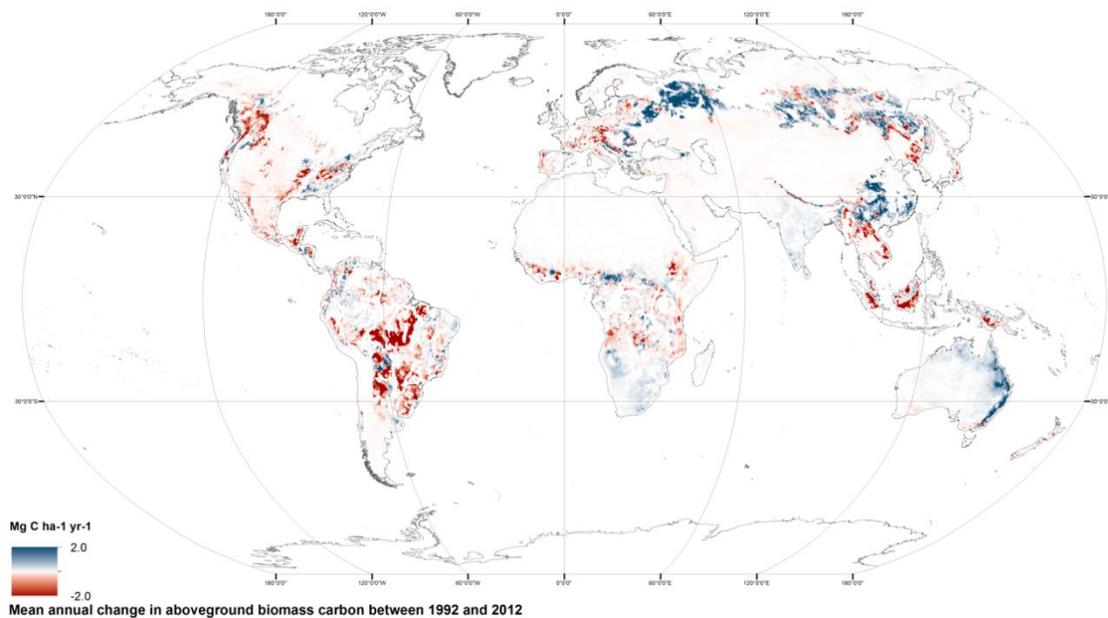
Figure 2.9. Mean annual change in aboveground biomass from 1993 to 2012; data from (Liu et al. 2015).

The indicators relating to forest structure suggest that deforestation has gone beyond the precautionary 'safe limit' for land-system change proposed in the Planetary Boundaries framework (Steffen et al. 2015b). That framework argues that reduction of forests below 75% of their natural extent risks dangerous reduction in biotic regulation of global climate, though there is uncertainty over exactly where the danger point lies (Steffen et al. 2015b). The global area of *tree cover* (assessed from remote-sensing data (Song et al. 2018)) is estimated to be only 54.2% of the area at the dawn of human civilisation, while current *extent of forests* (defined as having tree cover > 10%, aggregated from national statistics (FAO 2016a)) is 68.1% of their pre-industrial extent. These values are 1250 million ha and 460 million ha, respectively, below the proposed safe limit; as a comparison, Brazil's area is 852 million ha.

Deforestation has slowed since its peak in the 1990s. The *extent of forests* fell markedly more slowly in 2005-2015 than in 1990-2005 (FAO 2016a), and global *tree cover* has actually risen, by 2.6% per decade from 1982-2016 (Song et al. 2018). However, both indicators are still falling in the tropics while rising in temperate and boreal regions (FAO 2016b; Song et al. 2018); and approximately 15.3 billion trees are still being lost each year, through deforestation, forest management, disturbance and land-use change (Crowther et al. 2015).

The rapid increase in *leaf area index* that is apparent (Figure 2.8) (the area of leaves per unit area of land) is largely driven by changes in north temperate latitudes where climate change has

increased annual plant growth (Zhu et al. 2013). Mechanistic models (Le Quere et al. 2018) infer that global *soil organic carbon* (see Figure 2.8B) now stands at 104% of the level in the 1860s; but an alternative correlative approach estimates that land use has reduced levels to 92% of their natural baseline (Van der Esch et al. 2017). These diverging estimates could be partly reconciled if much of the loss caused by land-use change was before 1860; but more observation and modelling are needed.

For the indicators where we were able to make the comparison, ecosystem structure is on average less intact and declining more rapidly in the terrestrial hotspots of species rarity (as demarcated in Section 2.2.3.4.2) than across the world as a whole. Only 35.2% of their land area is still classed as *primary vegetation* and per-decade loss has averaged -5.1% of the 1970 level (the global figures are 39% and -4.1%, respectively). The corresponding values for *land neither cultivated nor urban* (ESA 2017) in hotspots (71.7% and –0.6% per decade) are also worse than across the world as a whole (76.7% and -0.2%, respectively: Figure 2.10). The habitat integrity (*Biodiversity Habitat Index* (Hoskins et al. 2018)) of these rarity hotspots is only 58%, much less than the overall global estimate of 70%.

By contrast, ecosystem structure is on average more intact and declining more slowly in Indigenous lands than across the world as a whole. Nearly 50% of mapped Indigenous land (Garnett et al. 2018) is still *primary vegetation* (Hurtt et al. 2018); and the rate of decline is only –2.8% per decade. Likewise, 93.2% of Indigenous Land (Garnett et al. 2018) is *neither cultivated nor urban* (ESA 2017), and this fraction is declining only a third as rapidly in Indigenous lands as it is globally (-0.2% vs -0.6% per decade).
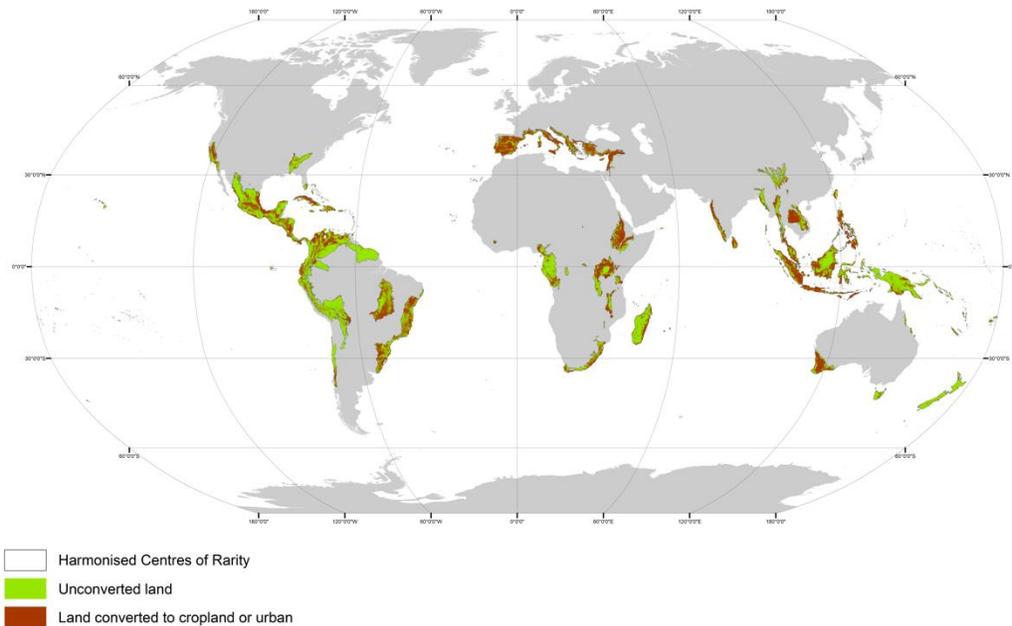
Figure 2.10. Many terrestrial hotspots of endemic species (harmonized across multiple taxonomic groups as in Figure 2.3, Section 2.2.3.4.2) have experienced widespread conversion of natural habitat to cropland and urban areas, according to satellite-derived land-cover data (ESA 2017)

Knowledge gaps: There are few indicators for the structure of freshwater or marine ecosystems, especially in the deep sea. Ecosystem condition is less well represented than ecosystem extent (because it is harder to measure consistently across space and over time), meaning that important degradation of ecosystem structure may be missed. For example, an estimated 35.9 Pg of soil was lost to erosion in 2012, 2.5% more than in 2001 (Borrelli et al. 2017), with soil eroding from conventional agricultural landscapes far more rapidly than it is formed (FAO & ITPS 2015). Land degradation – of which soil erosion is but one facet – is a global problem, affecting all land systems in all countries, but there is no quantitative consensus on its extent or trend (IPBES 2018): e.g., estimates of the still undegraded fraction of the land surface range from 75.8% to 96.8% (Gibbs & Salmon 2015). Estimates of the current global extent of grazing land also vary widely (Prestele et al. 2016, Phelps & Kaplan 2017).

2.2.5.2.2 Ecosystem function

(N.B. Italics denote indicators plotted in Figure 2.11.)

Evidence suggests that rates of some fundamental ecosystem processes have accelerated greatly (Figure 2.11). For example, the terrestrial *biomass turnover rate* – how quickly biomass is broken down and replaced – has nearly doubled on average; has increased more than tenfold in

croplands and artificial grasslands; and has increase at least threefold in east and south Asia and western, eastern and southern Europe (Erb et al. 2016).

Two differently-estimated indicators of *terrestrial Net Primary Production (NPP)* – which forms the base of most ecological food webs and material NCP – suggest slightly different trends. An ensemble of process-based models (Le Quere et al. 2018) suggests terrestrial NPP has risen by 2.6% per decade since 1970 – though the trend is flat over the past decade – and is now nearly 30% higher than in the 1860s (the earliest decade modelled). These models all assume that rising atmospheric CO_2 boosts photosynthesis, but the magnitude of this CO_2 fertilization effect is highly uncertain (Wenzel et al. 2016). In contrast, estimates derived instead from satellite data (Zhao & Running 2010) suggest a less rapid (and not statistically significant) increase, over the much shorter time period for which the data are available (Wang et al. 2012). The approaches agree, however, that the overall change masks wide spatial heterogeneity in the trend (Figure 2.12, Zhao & Running 2010). *Marine NPP* (Behrenfeld & Falkowski 1997) rose by 4.7% from 1998-2007.
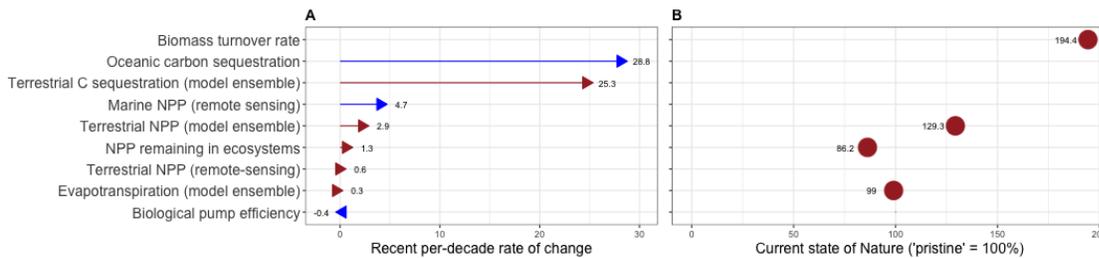
Figure 2.11: Global indicators of rates of ecosystem function. Marine indicators are in blue, terrestrial in brown. **A**: Trends, shown as the average per-decade rate of change since 1970 (or since the earliest post-1970 year for which data are available), ordered by rate of change; seven of the 8 global indicators suggest rates have been increasing (right-pointing arrows). **B**: Estimated current status relative to a pristine or at least largely pre-industrial baseline. Some indicators provide only either rate or status so appear in only one panel. See Supplementary Material for Chapter 2 Nature for detailed information on each indicator and its trend.

Carbon sequestration from the atmosphere helps to slow climate change, making it another important ecosystem function to measure. The ensemble of process-based models suggest *terrestrial carbon sequestration* has recently been rising by 25% per decade and *oceanic carbon sequestration* by 29% per decade (Le Quere et al. 2018), despite a slight reduction in the *efficiency of the biological pump* (Cael et al. 2017).

The annual amount of *NPP remaining in terrestrial ecosystems* after human appropriation (Krausmann et al. 2013) is now around 86% its inferred natural baseline level (though only 64% in Asia). Its slow net change through history probably reflects a near-balance between increasing human appropriation of NPP and increasing NPP caused by land management and (increasingly in recent decades) CO_2 fertilisation (Krausmann et al. 2013). However, the biotic consequences could be much greater than such a small net change might suggest: agriculture has increasingly

channelled terrestrial NPP through a relatively small set of species, reducing the diversity of forms in which that NPP is available to the species in ecosystems.

> +20

0

< -20

Global Net Primary Productivity Trend 2003 - 2015 (gC/m2/year2)
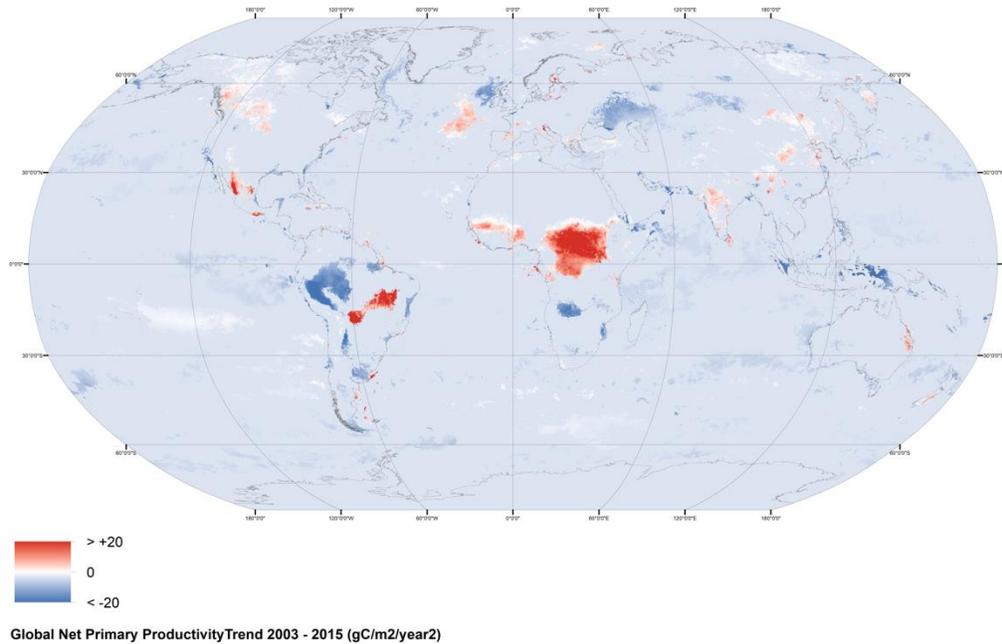
Figure 2.12 Spatial variation in the trend in terrestrial and marine NPP from 2003-2015, estimated from remote sensing data (terrestrial – Zhao & Running 2010; marine – Berhrenfeld & Falkowski 1997). Note that the spatial pattern has itself changed over time, so may be different in other time windows.

Knowledge gaps

Ecological communities carry out many more ecosystem functions vital for ecosystem health and the delivery of NCP, such as pollination, decomposition, fruit and seed dispersal, pest control and fertilization of the soil (Diaz et al. 2018, Chapter 2.3); however, available indicators mostly report on either the status of the species responsible or the NCP, rather than on the ecosystem functions and processes linking the two. This partly reflects the difficulties of scaling from local sites, where ecosystem function can be measured, to the globe. More global indicators are needed of rates of ecosystem processes that directly underpin particular NCP or that indirectly underpin ecosystem health.

2.2.5.2.3 Community composition

(N.B. Italics denote indicators plotted in Figure 2.13.)

Local communities are not on average showing rapid changes in species-richness, but their biotic integrity is being eroded rapidly by changes in which species are present and abundant (Figure

2.13, blue background). Local assemblages are also becoming more similar to each other, a pattern known as biotic homogenization. At regional scales, the numbers of species – especially non-native species – have tended to increase over recent decades (Figure 2.13, orange background).

a. Composition of local communities

The average balance between gains and losses of species in local assemblages worldwide remains unclear (Cardinale et al. 2018), largely because rates of gain (of alien, disturbance-tolerant or other human-adapted species, or of climate migrants) and of loss (though local extinction) are very context-dependent (e.g., Thomas 2013). The *BioTime species-richness* indicator, estimated as the average trend from a compilation of time-series data from local terrestrial, freshwater and marine assemblages around the world (Dornelas et al. 2014), shows a slight but not statistically significant increase on average with very wide variation from site to site (Dornelas et al. 2014). A compilation of coastal marine assemblages tended to gain species over time, but sites facing local human impacts tended to lose species, especially rare species (Elahi et al. 2015); and a set of local plant communities showed an average decrease in species richness in the tropics but an increase in north temperate regions (Vellend et al. 2013) – assemblages facing disturbance tend to lose species whereas those recovering after disturbance tend to show gains (Gonzalez et al. 2016). Geographic biases in such collations mean they may not accurately reflect the widespread increase in drivers over recent decades (Elahi et al. 2015, Gonzalez et al. 2016). The *PREDICTS species-richness* indicator (Hill et al. 2018), which tries to overcome such geographic biases using a statistical model, shows a slight decrease over time; but the statistical model does not incorporate effects of alien species (Newbold et al. 2015).
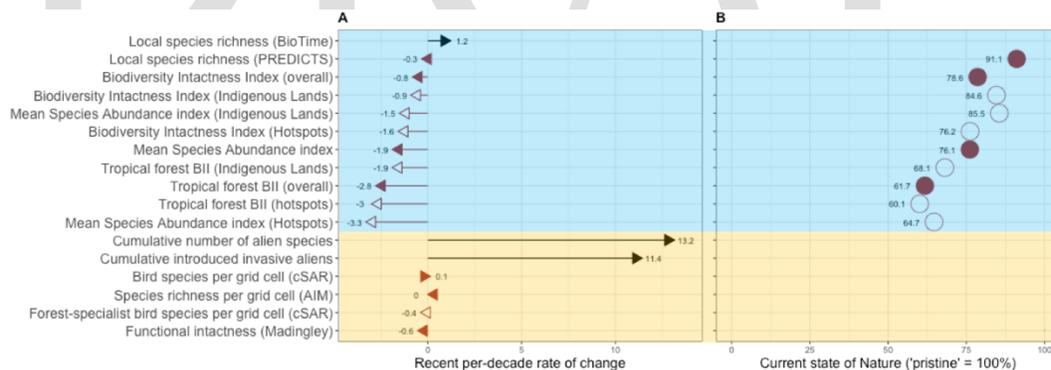
Figure 2.13: Global indicators of community composition at the local scale (blue background) and the regional scale (orange background). Brown symbols are terrestrial indicators, black symbols are indicators that combine terrestrial, freshwater and marine data. Solid symbols represent overall values for indicators, whereas semi-transparent points represent values for subsets (e.g., for a particular biome or functional group) of the overall indicator. **A**: Trends, shown as the average per-decade rate of change since 1970 (or since the earliest post-1970 year for which data are available), ordered by rate of change. **B**: Estimated current status relative to a pristine or at largely pre-industrial baseline. Some indicators provide only either rate or status so appear in only one panel. Supplementary Material for Chapter 2 Nature has detailed information and full references for each indicator, including subsets.

Two indicators – *Biodiversity Intactness Index* (BII: Hill et al. 2018, De Palma et al. 2018) and *Mean Species Abundance* (Schipper et al. 2016) – agree that biotic integrity has declined on average to well below its proposed safe limit in the Planetary Boundaries scheme (Steffen et al. 2015b). That framework suggests that large regions whose biotic integrity – i.e., the fraction of originally-present biodiversity that remains – falls below 90% risk large-scale failure of ecosystem resilience that would cause critical reductions in the flows of nature's contributions to people (Steffen et al. 2015b) though there is a great deal of uncertainty about precisely where any boundary should be placed (Steffen et al. 2015b, Mace et al. 2014). A global model (Hill et al. 2018) estimates the *Biodiversity Intactness Index* (BII) to average only 79% across terrestrial ecosystems (Figure 2.14), with most biomes below 90%; a model focused on tropical and subtropical forest biomes (De Palma et al. 2018) estimates an even lower BII and more negative trend, as does the global model of *Mean Species Abundance* (Schipper et al. 2016). For both BII indicators and Mean Species Abundance, hotspots of rare and endemic species have a lower current status and a more negative trend than the global average, whereas Indigenous lands have a better current status (though still below the proposed Planetary Boundary) and usually a slower rate of decline (Figure 2.13).
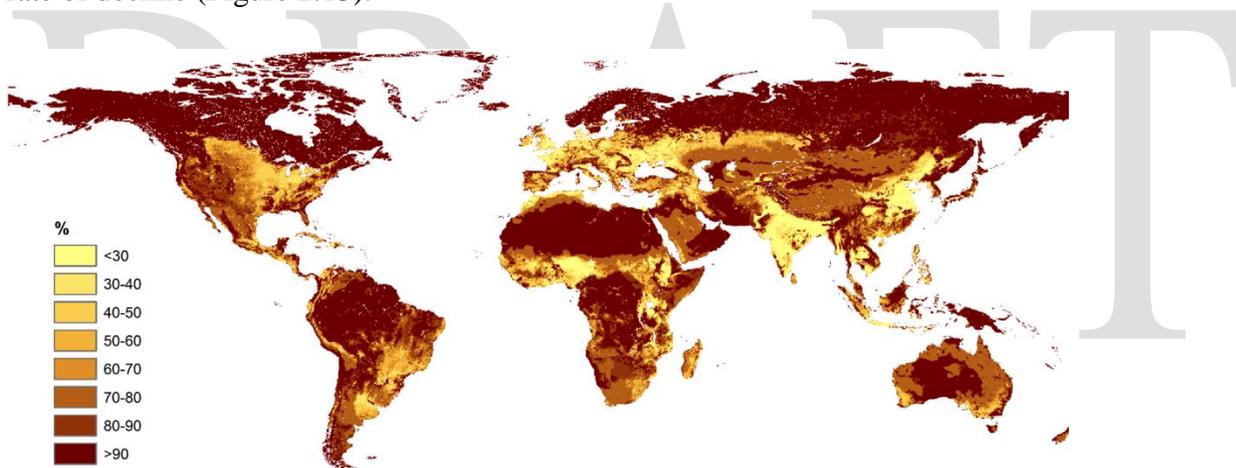
Figure 2.14_COMPOSITION: Global map of estimated terrestrial Biodiversity Intactness Index in 2015 (Hill et al. 2018). Darker colours indicate more intact ecological community composition.

b. Compositional dissimilarity between assemblages
Local assemblages are becoming more similar to each other on average, a phenomenon termed biotic homogenization (McKinney & Lockwood 1999) or the 'anthropogenic blender' (Olden 2006). When human actions add species to a local assemblage, they are often likely to add the same species to many other assemblages within the region or even around the world; e.g., we plant and farm a relatively small number of species over vast areas of land. The structural, chemical and biotic sameness of these anthromes means that species adapted to them – whether alien or native – can spread widely. Shipping transports ballast water, and its complement of species, from one harbour to another. We move the same pets, pests, pathogens and ornamental species around the world. All of these additions are likely to make the assemblages more similar.

At the same time, the species lost from local assemblages because of human actions often differ from place to place, in which case their loss also makes assemblages more similar. A global synthesis reported significant homogenisation across nearly all taxonomic groups at nearly all scales (Baiser et al. 2012); further support comes from regional syntheses (e.g., Rahel 2000, Winter et al. 2009; Solar et al. 2015) and the most detailed field studies (e.g., Gossner et al. 2016).

c. Composition of regional assemblages

Numbers of species in assemblages at larger spatial scales – such as countries or 0.25^{o} grid cells – have tended to increase over recent decades (Figure 2.13, orange background), partly driven by rapid increases in numbers of non-native species (Thomas 2013, McGill et al. 2015). A global analysis of establishment of species in new countries from a wide range of taxonomic groups found the *cumulative number of alien species* is rising by 13% per decade, with 37% of all reported establishment events being since 1970 (Seebens et al. 2017). Across 21 countries with particularly good recording of *introduced invasive alien species* (i.e., aliens that cause ecological or economic problems), numbers per country have increased by an average of 70% since 1970 (Pagad et al. 2015). Among the most widespread invaders are the black rat (*Rattus rattus*, 23% of the world's countries), water hyacinth (*Eichhornia crassipes*, 30%), Eastern mosquitofish (*Gambusia holbrooki*, 30%), purple nutsedge (*Cyperus rotundus*, 37%), and cottony cushion scale insects (*Icerya purchasi*, 42%) (Turbelin et al. 2017). Many crop pests and pathogens – especially fungal pathogens – have become widespread, tracking the regional expansion of their host crops (Bebber et al. 2014).

Over 13,000 plant species of plant have become established in countries outside their native range (van Kleunen et al. 2015). Numbers of plant species have increased by an average of 20%-25% across continental regions in Europe and the USA because establishment of aliens has exceeded losses of natives at this scale (Vellend et al. 2017); regional plant species richness is estimated to have increased by 5% or more across nearly half of the world's land surface and decreased similarly across only 14% (Ellis et al. 2012). Alien species make up a smaller fraction of the flora in tropical countries than in temperate ones, but too little is known about national extinctions in the tropics to be sure that the net change there has been an increase (Vellend et al. 2017). *Species richness per grid cell* (Kim et al. 2018)*,* modelled across plants, birds, mammals, amphibians and reptiles, has fallen slightly since 1970 because of changes in land use and climate. However, this model omits species introductions (Kim et al. 2018), which would make the trend more positive; and, even without introductions, the indicator is still higher than in 1900 for most groups. A conceptually similar model (Kim et al. 2018; Pereira & Daily 2006) estimates that *bird species per grid cell* has risen slightly since 1970, but that *forest-specialist bird species per grid cell* has fallen, and more steeply. A mechanistic general ecosystem model (Harfoot et al. 2014) suggests that average *functional intactness* (i.e., the extent to which a region's species still occupy the functional trait space of its native species) is falling because of harvesting of primary productivity and climate change (Kim et al. 2018).

Island assemblages are likely to be an exception to the general trend towards increased species numbers. They can be devastated by invasive alien species (e.g., O'Dowd et al. 2003, Bergstrom et al. 2009, Reaser et al. 2007), in part because native species may have evolved in the absence of strong competition, predation or pathogens (Courchamp et al. 2003). Introduced mammalian predators have removed many native bird species from oceanic islands worldwide (Blackburn et al. 2004), reducing diversity at the island scale. Introduced plant species, by contrast, have roughly doubled the numbers of plant species on a set of well-documented oceanic islands (Sax & Gaines 2008, Carvallo & Castro 2017). Even though they may increase regional diversity, though, invasive alien plants usually reduce numbers of species in local assemblages on islands (Pysek et al. 2012) and can have profound ecosystem impacts (e.g., Dulloo et al. 2002, Tassin et al. 2006, Pysek et al. 2012).

Some invasive alien species on mainlands can also drive reductions in regional-scale diversity, by causing native species to decline. *Batrachochytrium dendrobatidis*, an infectious fungal pathogen that has infected over 700 amphibian species worldwide, has caused a number of extinctions, and is recognised as a threat to nearly 400 species (Lips 2016, Olson et al. 2013, Bellard et al. 2016)

Even where regional species-richness has increased, the increase may be temporary because an 'extinction debt' has not yet been repaid (Jackson & Sax 2010). Biotic responses to drivers of change are often not immediate, meaning recent intensification of any driver can produce 'dead species walking', certain to disappear from the region unless the drivers of their decline are reversed (Kuussari et al. 2009). Extinction debts are discussed in more detail in Section 2.2.5.2.4a below.

Knowledge gaps: Available indicators all relate to the taxonomic or functional composition rather than the interactions among organisms and taxa. Indicators overwhelmingly relate to terrestrial free-living animal and plant species: freshwater and marine assemblages are greatly under-represented, and microbial and parasite assemblages entirely so. As yet there are no global indicators of biotic homogenization.

2.2.5.2.4 Species populations

(N.B. Italics denote indicators plotted in Figure 2.16.)

a. Extinctions, extinction risk and extinction debt
The most direct evidence on global extinctions and extinction risk comes from the detailed assessments of species' conservation status undertaken by the IUCN (International Union for the Conservation of Nature). IUCN has assessed the global conservation status of 93,579 species, mostly vertebrates, of which 872 (0.9%) have gone extinct since 1500 (IUCN 2018). Under-recording and time lags in recognizing extinction events make this a certain underestimate of the true number (Dunn 2005, Pimm et al. 2006, Stork 2011, Alroy 2015, Scheffers et al. 2012), especially in less well studied groups (e.g., only 62 species of insect are listed as extinct; but fewer than 1% of insects have been assessed: IUCN 2018) and habitats (e.g., only 20 marine

extinctions have been recorded (Webb & Mindel 2015)). In the best-recorded groups – mammals and birds – around 1.4% of species are known to have gone globally extinct since 1500, most of them since 1875 (IUCN 2018).

The global rate of species extinction is already at least tens to hundreds of times higher than the average rate over the past 10 million years, and is accelerating (Barnosky et al. 2010, Pimm et al. 2014, Ceballos et al. 2015); the difficulties of estimating and comparing current and past extinction rates (Barnosky et al. 2010, Pimm et al. 2014, Ceballos et al. 2015) preclude greater precision. The extinction rate therefore already exceeds its proposed safe limit (set at ten times the average rate (Steffen et al. 2015b)) in the Planetary Boundaries framework, though the suggestion that elevated rates may eventually trigger sharp and irreversible changes in the Earth system (Steffen et al. 2015b) has been criticised (Mace et al. 2014; Brook et al. 2013). Extinction rates would be still higher but for successful conservation (Butchart et al. 2006, Chapter 3).

Extrapolating from detailed assessments of species across a growing and diverse set of well-studied taxonomic groups, it is probable that at least a million animal and plant species – more than one in eight – are currently threatened with global extinction. The proportion of species currently threatened with global extinction (i.e., listed in the IUCN Red List as Vulnerable, Endangered or Critically Endangered) averages around 25% across a wide range of animal and plant taxonomic groups (range = 7.4%-63.2%, median = 22.1%; Table 2.1). The current prevalence of extinction risk appears to be similar between terrestrial and marine realms, from the few marine groups in Table 2.1 and from models of how threat prevalence scales with the comprehensiveness of Red List assessments (Webb & Mindel 2015). No global estimate of extinction risk prevalence is yet available for any of the hyperdiverse insect orders. However, a cautious estimate of 10% is reasonable, based on the Red Lists for Europe (the region with the best data), which report that 9.2% of bee species (Nieto et al. 2014), 8.6% of butterflies (van Swaay et al. 2010) and 17.9% of saproxylic beetles (Calix et al. 2018) are threatened with regional extinction. (For context, in vertebrates, Europe's levels of regional extinction risk are lower than the overall levels of global extinction risk (EU 2018)). If insects make up three quarters of animal and plant species (Chapman 2009) and only 10% of them are threatened as opposed to 25% of species in other groups, then overall nearly 14% of animal and plant species are threatened with extinction, i.e., more than a million using the estimated total number of 8.1 million (Mora et al. 2011).

Table 2.1 Proportions of evaluated species that are threatened with extinction, according to IUCN Red List assessments, within different taxonomic groups. The first figure given assumes that Data Deficient species are equally likely as other species to be threatened. The range reported shows the proportion if Data Deficient species are assumed to be not threatened and threatened, respectively. Basis of estimate: all species = comprehensive assessment of whole group; sample = representative sample assessed; some families = all species within some families assessed, but families may not be representative.

Group	Threatened species (%)	Possible range (%)	Basis of estimate	Reference
Vertebrates				
Amphibians	41.49%	32-55%	all species	IUCN (2018)
Birds	13.47%	13-14%	all species	IUCN (2018)
Bony fishes	7.41%	7-18%	some families	IUCN (2018)
Mammals	25.17%	22-36%	all species	IUCN (2018)
marine mammals	38.70%	30-52%	marine species	
Reptiles	18.99%	15-36%	sample	Bohm et al. (2013)
Sharks & rays	31.18%	18-60%	all species	IUCN (2018)
Invertebrates				
Crustaceans	27.49%	17-56%	some families	IUCN (2018)
Gastropods	7.52%	6-20%	some families	IUCN (2018)
Odonata	15.38%	10-45%	sample	Clausnitzer et al. (2009)
Reef-forming corals	32.91%	27-44%	all species	IUCN (2018)
Plants				
Cycads	63.16%	63-64%	all species	IUCN (2018)
Dicots	36.14%	32-44%	some families	IUCN (2018)
Legumes	11.30%	11-18%	sample	Brummitt et al. (2015)
Gymnosperms	40.55%	40-42%	sample	Brummitt et al. (2015)
Monocots	17.51%	15-27%	sample	Brummitt et al. (2015)
Pteridophytes	16.01%	15.9-16.4%	sample	Brummitt et al. (2015)

Numbers of threatened vertebrate species show wide geographic variation both on land and at sea (Figure 2.15), reflecting where large numbers of narrowly-distributed species (see Section 2.2.3.4.2) face often intense, often multiple anthropogenic drivers (Hoffmann et al. 2010).

The *Red List Index* (RLI) (Butchart et al. 2007; Butchart et al. 2010) tracks overall trends in survival probability (the inverse of extinction risk) of species in taxonomic groups whose IUCN Red List status has been assessed multiple times. Overall, the RLI is now only 75% of the value it would have without human impacts (Figure 2.16), though this varies among taxonomic groups (e.g., birds have an RLI around 90% but for cycads RLI is below 60%: Chapter 3). Regions showing the greatest deterioration in RLI include much of Southeast Asia and Central America (Hoffmann et al. 2010). RLI values calculated for sets of species that directly deliver some NCPs – *internationally-traded species*, *pollinating vertebrate species*, *species used in food and medicine*, and *wild relatives* of farmed and domesticated mammals and birds – are higher than the overall value and are declining more slowly, but they are all declining. Species' progress towards extinction appears to be increasingly rapid: half of the decline in the overall Red List Index has taken place in the last 40 years.

Few insects have global IUCN assessments, but regional and national assessments of insect pollinators often indicate high levels of threat – often more than 40% of species threatened at a national scale – particularly for bees and butterflies (IPBES 2016). Recent European scale assessments indicate that 9.2% of bees (Nieto et al. 2014) and 8.6% of butterflies (van Swaay et al. 2010) are threatened. Bee species that pollinate crops are generally common with a low prevalence of extinction risk (IPBES 2016).

Threatened species richness

Number of comprehensively assessed species with threat status 'Critically Endangered', 'Endangered' and 'Vulnerable' potentially occurring in each ~10km² grid cell.

Marine			Terrestrial		
High: 393		Low: 1	High: 58		Low: 1

Coordinate System Geographic

Data sources: IUCN Red List of Threatened Species (2017) Version 2017.1. http://www.iucnredlist.org.
Methods: For Threatened species richnes, each 10km grid cell was scored for richness for each species inIUCN categorie 'Critically Endangered', 'Engangered' or 'Vulnerable' (i.e each species scored a 1 in a cell if the range overlapped with the cell) and given a total score by summing scores across all the species potentially occurring in it. The data covered species from all taxonomic groups that have been comprehensively assessed resulting in a database of 23,442 species in the terrestrial realm and 6,812 in the marine realm from which the Critically Endangered , Endangered and Vulnerable species were selected. The following taxonomic groups were included **Terrestrial species groups:** Amphibians, Birds, Chamaeleonida (Chameleons,Crocodylia (Crocodiles/Alligators) and Mammals. **Marine species groups:** Angelfish, Birds, Blennies, Bonefish and tarpons, Butterflyfish, Cheloniidae (Marine Turtles), Chondrichthyes (Sharks and Rays), Conus (Cone Snails), Corals, Damselfish, Groupers, Hagfish, Lobsters, Mammals, Mangroves, Seabreams and porgies, Pufferfish, Sea cucumbers, Seagrasses, Sea snakes, Sturgeonfish, tangs and unicornfish, Tunas and billfishes and Wrasse. Although all IUCN Red List species range data for comprehensively assessed taxonomic groups have been included in the analysis, these represent only part of all biodiversity known to occur globally, and are biased towards vertebrates.

The boundaries and names shown and the designations used on maps do not imply official endorsement or acceptance by UN Environment or contributory organisations.

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Figure 2.15. Numbers of threatened (i.e., vulnerable, endangered or critically endangered) species per 10km grid cell, pooled from comprehensive geographic distribution and extinction-risk assessments of multiple taxonomic groups. Green = terrestrial (amphibians, birds, chameleons, crocodiles/alligators and mammals); blue = marine (angelfish, birds, blennies, bonefish/tarpons, butterflyfish, marine turtles, sharks/rays, *Conus* cone shells, corals, damselfish, groupers, hagfish, lobsters, mammals, mangroves/seabreams/porgies, pufferfish, sea cucumbers, seagrasses, sea snakes, sturgeonfish/tangs/unicornfish, tunas/billfishes and wrasse; grey = no data. Darker colours indicate higher numbers of threatened species. Note that only a small minority of taxonomic groups have so far been assessed, with a bias towards vertebrates especially on land. Methods as in Hoffmann et al. (2010). Figure produced by UNEP-WCMC.

Whereas IUCN's detailed Red List assessments of species form the basis for 'bottom-up' estimates of numbers of threatened species, an alternative 'top-down' approach can be used to estimate the 'extinction debt' – i.e., how many species are expected to eventually go extinct because of habitat deterioration that has already taken place (Kuussari et al. 2009). The earliest estimates of extinction debt (Diamond 1972) were based directly on one of the strongest patterns in biodiversity, the species-area relationship: the number of species in a region increases predictably with its area (often as a power law), because larger regions both have greater habitat diversity and can support larger numbers of viable populations (Rosenzweig 1995, Lewis 2006). Habitat loss effectively makes the region smaller. Though this loss of area may not wipe any species out immediately, it means that the region now has more species than expected: this excess of species is the extinction debt, and all the region's species will have elevated probabilities of extinction until the diversity falls back to the level expected from the species-

area

1.1.1 Box 2.3: The contemporary globalization of native Amazonian and American plants

Fait avec Philcarto * Bastien Beaufort 2018 * http://philcarto.free.fr

61 283 408	Chine
25 109 000	Inde
5 911 681	Etats-Unis d'Amérique
2 409 160	Ghana
814 916	Algérie

Legend:
- Tobacco
- Capsicum peppers
- Papaya
- Beans
- Pumpkins, Calabashes & squashes
- Cocoa
- Avocado
- Pineapples
- Peanuts
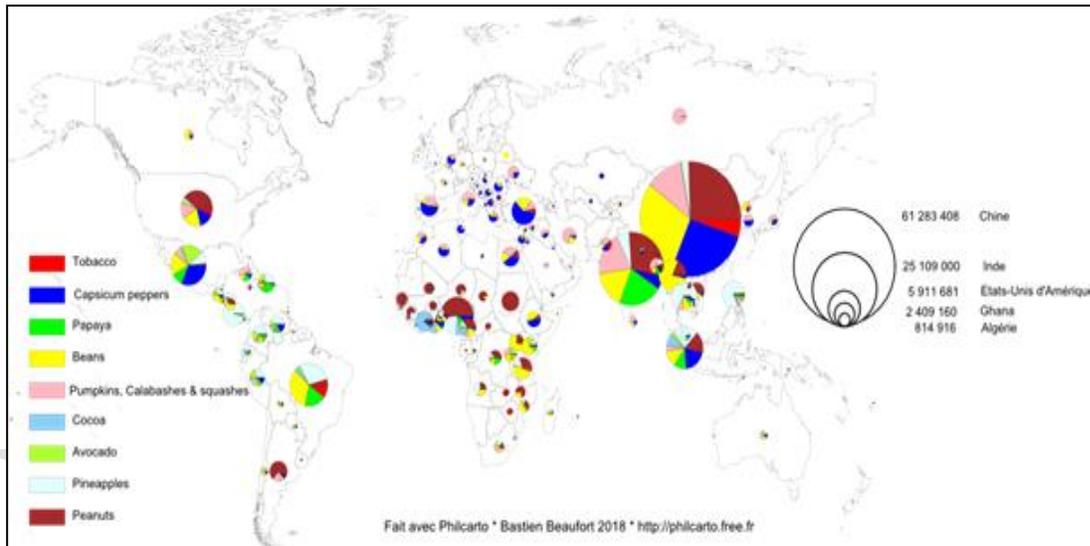
This map shows the current global centres of production (in tons) of key crops that originated from native American and Amazonian plants (Beaufort 2017). Some important Amazonian crops, such as manioc and rubber, are not displayed.

The map highlights that many crops originating from agrobiodiverse regions are now used well beyond their centres of origin and domestication; and that the Amazon – often portrayed as the ultimate example of "pristine forest" – is actually a hugely important centre of domesticated nature, contributing significantly to the global agricultural economy.

One of the most globally widespread domesticated Amazonian plant genera is *Capsicum* (pepper; species *annum, chinense*, and *pubescens*). Other examples from the Amazon include pineapple (*Ananas comosus*), papaya (*Carica papaya*) and peanuts (*Arachis hypogeae*), which originated in South-West Amazon rainforest. Cocoa is also another globally important plant, which has at least ten different domesticated indigenous varieties scattered across the Amazon rainforest. Many of these cocoa varieties, as with dozens of other varieties of seeds and cultivars, are still managed by local traditional and indigenous groups in the Amazon. Sources: Beaufort 2017; FAOSTats 2014. Designed with PhilCarto.

relationship. Such approaches do not identify precisely which species in the region will go extinct; they may not meet IUCN's criteria for being listed as threatened, for example. Nor do these approaches specify how long the extinctions will take: although the first extinctions may arrive quickly, the last ones may take centuries, especially in large regions and/or when species

have long generation times (Kuussari et al. 2009; Vellend et al. 2006; Halley et al. 2016). The estimates of extinction debt used here come from models with more sophisticated species-area relationships that consider species' habitat preferences and geographic distributions, and habitat condition as well as extent (Pereira and Daily 2006, Hoskins et al. 2018, Kim et al. 2018), meaning many of the criticisms of earlier approaches (Lewis 2006, Hé & Hubbell 2011, Pereira et al. 2012) no longer apply. Furthermore, they use entirely different data and methods from the Red List assessments, so provide a completely independent line of evidence.

The most comprehensive global estimate available (Hoskins et al. 2018) suggests that the terrestrial extinction debt currently stands at hundreds of thousands of animal and plant species. The loss of terrestrial habitat integrity estimated by the *Biodiversity Habitat Index* (Hoskins et al. 2018), when coupled with the species-area relationship, suggests that only 92.1% of terrestrial vertebrate species, 91.6% of terrestrial invertebrates and 90.7% of terrestrial plants have sufficient habitat to persist. The numbers of plant and especially animal species remain very uncertain (Caley et al. 2014, Scheffers et al. 2012), but a recent non-extreme estimate of 8.1 million of which 2.2 million are marine (Mora et al. 2011), these proportions suggest that around half a million terrestrial animal and plant species are 'dead species walking', committed to extinction unless their habitats improve in time to prevent it. This total includes over 3000 vertebrates and over 40,000 plants. Even this estimate may be conservative, as undocumented diversity of arthropods, parasites and soil microfauna could mean there are 2-25 times more animal species than assumed here (Larsen et al. 2017), and fungi are not included (Scheffers et al. 2012). Using a related approach, the countryside species-area relationship (cSAR), to estimate the *global bird richness* that can persist suggests that 97.6% of the world's bird species, but only 94.9% of forest-specialist birds, will avoid extinction resulting from past habitat loss.

These two very different lines of evidence both point to a further sharp acceleration in the global rate of species extinction – already at least tens to hundreds of times higher than the average rate over the past 10 million years. The numbers of threatened species that will go extinct if the drivers that threaten them continue, and the numbers of 'dead species walking' that will die out even without any further habitat deterioration or loss, dwarf the numbers of species already driven extinct by human actions (Johnson et al. 2017; Wearn et al. 2012). Rapid large-scale restoration of habitats can pardon the 'dead species walking', provided it takes place in time (Kuussari et al. 2009); and even much less widespread restoration can greatly delay extinctions if targeted optimally (e.g., Newmark et al. 2017).

b. Geographic distribution and population size
Nearly all global indicators of geographic distribution (Figure 2.16, blue background) and population size (Figure 2.16, cream background) show rapid decline, reflecting widespread reductions in animal populations on land (Dirzo et al. 2014, Ceballos et al. 2017) and sea (McCauley et al. 2015), though most global indicators focus on vertebrates. Several indicators are calculated in a way that makes them particularly sensitive to trends in rare species (Buckland et al. 2011), and these all show rapid declines: *The Living Planet Index* (LPI) for vertebrate populations (McRae et al. 2017); the *Wild Bird Index* for habitat-specialist birds; and the *extent*

of suitable habitat for terrestrial mammals (Visconti et al. 2015; Kim et al. 2018). The *Species Habitat Index*, which changes in direct proportion to average species range size (Map of Life 2018), has shown more modest recent declines in terrestrial vertebrates. *Mammalian range size* has been reduced to an average of 83% of species' inferred original ranges, but *megafaunal range size* – species larger than 44.5kg – is now only 28% of the natural baseline (Faurby & Svenning 2015), with large mammal ranges having declined particularly rapidly in south and southeast Asia (Ceballos et al. 2017). *Predatory fish biomass* (which includes the main target species for fisheries (Christensen et al. 2014)) has been falling by -14% per decade, and the proportion of *fish stocks within biologically sustainable levels* by 6% per decade (to less than 70%) (FAO 2016c). The *biomass of prey fish* (Figure 2.16) has been rising by 10% per decade – the only indicator to show an increase – probably because fishing has removed predatory fish (Christensen et al. 2014). Such indirect responses to anthropogenic drivers are ubiquitous and can have profound effects on many aspects of ecosystems (Dirzo et al. 2014; McCauley et al. 2015).

Invertebrate trends have not so far been synthesised globally, because of a dearth of tropical data. An LPI-like analysis of mainly European and North American data reported a decline of -11% per decade (Dirzo et al. 2014). The same regions have seen declines in geographic distribution and occurrence of many wild bees and butterflies (IPBES 2016); and, of species with enough information to make an assessment, 37% of bees and 31% of butterflies are declining in Europe (Nieto et al. 2014, van Swaay et al. 2010, IPBES 2016). Available time-series data show that local declines of insects can be rapid even in the absence of large-scale land-use change (e.g., 76% decline over 27 years in biomass of flying insects in sites in 63 protected areas in Germany (Hallmann et al. 2017)); it is not known how widespread such rapid declines are.

Although many species are declining, farmed species, domesticates, and species that are well adapted to anthromes have all increased in abundance. A hectare of wheat will often have more than 500,000 established plants – and wheat is planted on around 220 million ha each year (Rudel et al. 2009); the number of managed western honey bee hives is increasing globally (IPBES 2016); and livestock now accounts for over 90% of megafaunal biomass on land (Barnosky 2008).

Knowledge gaps: There are shortages of detailed knowledge of conservation status and population trends in insect, fungal and microbial species. Tropical populations are extremely under-represented in trend data.
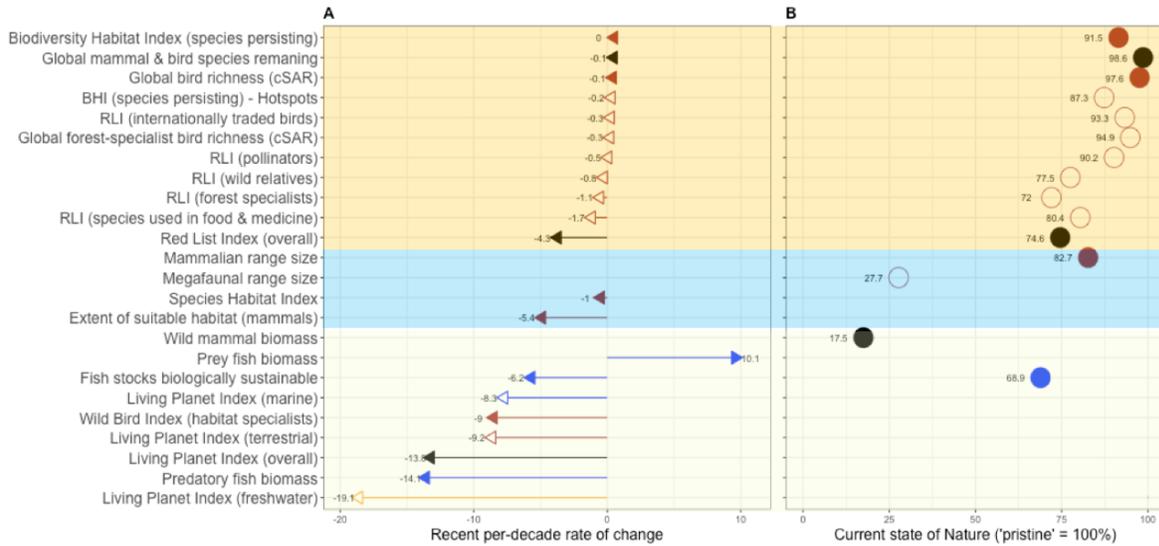
Figure 2.16: Global indicators of species population, reflecting persistence of species (orange background), geographic range size (blue background) or population size (cream background). Terrestrial indicators are shown in brown, marine in blue, freshwater in orange, and multi-realm indicators in black. Solid symbols represent overall values for indicators, whereas semi-transparent points represent values for subsets (e.g., within hotspots of endemic species) of the overall indicator. **A**: Trends, shown as the average per-decade rate of change since 1970 (or since the earliest post-1970 year for which data are available), ordered by rate of change. **B**: Estimated current status relative to a pristine or at largely pre-industrial baseline. Some indicators provide only either rate or status so appear in only one panel. Supplementary Material for Chapter 2 Nature has detailed information and full references for each indicator, including subsets.

2.2.5.2.5 Organismal traits

(N.B. Italics denote indicators plotted in Figure 2.18.)

Human activities have driven and continue to drive widespread changes in distributions of organismal traits within populations (Figure 2.17) and in local, regional, and global assemblages (Figure 2.18, Figure 2.19). Traits not only mediate how populations and communities respond to changing environments (e.g., Suding et al. 2008, Mouillot et al. 2013, Diaz et al. 2013, Jennings et al. 1998, Hevia et al. 2017) but also strongly influence species' likelihoods of being exploited (Jerolozimski & Peres 2003), persecuted (Inskip & Zimmermann 2009), domesticated (Larson & Fuller 2014), introduced (Theoharides & Dukes 2007) or otherwise impacted by people. Rapid evolution (Box 2.5) contributes to the changes, alongside phenotypic plasticity (in which the environment shapes how an organism's phenotype develops) and ecological processes. The combined effects typically shift both average trait values (e.g., toward smaller body size) and the amount of trait variation (e.g., reducing the range of trait values). The changes in trait distributions matter because they can have consequences – sometimes major – for ecosystem functioning, NCPs, and whether ecosystems will be resilient in the face of ongoing environmental change (Lavorel & Garnier 2002, Diaz et al. 2013, Laliberté et al. 2009).

Few quantitative indicators are available that show how distributions of organismal traits have changed globally, but there is an extensive literature showing how each of the main direct drivers affects both trait distributions among and within species. This section highlights some recent examples, while Box 2.5 focuses on within-population changes, especially heritable genetic changes – evolution.

Land-use change causes the assembly of new ecological communities, often with very different trait distributions from the community present previously. Forest removal obviously greatly changes distributions of plant traits, for instance, but also reshapes trait distributions in tropical bird assemblages: long-lived, large, non-migratory, forest-specialist frugivores and insectivores become less abundant and less widespread (Newbold et al. 2013). Increasing land use led to European plant communities being dominated by shorter species with more acquisitive leaf syndromes and accelerated flowering phenology (Garnier et al. 2007). Bee species' responses to changing land use in Europe depend on flight season duration, foraging range and, to a lesser extent, niche breadth, reproductive strategy and phenology (De Palma et al. 2015). A global meta-analysis found that intensification of land use was associated with greater reduction of functional diversity in mammal and bird assemblages than expected from the number of species lost (Flynn et al. 2008).

Direct exploitation often targets older, larger and more accessible individuals, so shifts trait distributions in the opposite direction. For example, large, diurnal, terrestrial mammals have been particularly likely to face hunting pressure (Johnson 2002), and species of tuna and their relatives that grow and reproduce more slowly have declined more than other species in the face of fishing pressure (Juan-Jordá et al. 2015). Such phenotype-dependent mortality holds both among populations within species (Darimont et al. 2009), so larger-bodied species are lost from communities, larger-bodied populations are lost from species, and many populations rapidly evolve smaller body size and earlier maturation (Box 2.5).

Climate change tends to shift trait distributions away from low reproductive rates, poor dispersal abilities and ecological specialism (as species with these traits are less able to persist when climate change: Pacifici et al. 2015) and towards more flexible – environmentally responsive – phenotypes (e.g., plants: Willis et al. 2008, birds: Both et al. 2006, Nussey et al. 2005) and earlier spring phenology in seasonal environments (e.g., earlier bud break for plants, earlier hatching and emergence for insects, and earlier breeding for birds and mammals (Parmesan and Yohe 2003; Wolkovich et al. 2012)). Global changes in phenology have been dramatic: between 1981 and 2012, the phenology of vegetation (timing of leaf onset and offset) has changed by more than 2 standard deviations across 54% of the global land surface (Buitenwerf et al. 2015), and growing seasons have lengthened (Linderholm 2006) – in the Arctic by more than 3 days per decade (Xu et al. 2013). This information is policy relevant because it can influence decisions about assisted migration (moving species to locations where they will be better suited for the new climate: McLachlan et al. 2007).

Pollution also reshapes trait distributions, in ways that differ among pollutants and species. Effects of different classes of insecticide on aquatic invertebrates, for example, are mediated by the body size, respiration type and degree of sclerotization of species, populations and individuals (Rico & Van den Brink 2015).

Invasive alien species can increase trait and functional diversity having different trait values from natives (Ordonez et al. 2010, Van Kleunen et al 2010, Hejda & de Bello 2013), but their trait-mediated effects on native species can also change overall trait distributions. A global meta-analysis of 198 studies found that invasive plants tend to reduce diversity and abundance of herbivorous and carnivorous animals but not detritivores or omnivores (Schirmel et al. 2015), thereby changing the trophic diversity of assemblages.

Indirect effects of drivers – knock-on effects – can also select against particular organismal traits and therefore affect trait distributions. Most obviously, species that depend on just one or a narrow set of other species – whether as a host, food, pollinator, or disperser – will often be vulnerable if that species declines (Dunn et al. 2009).

Species' extinction risk – which integrates across all direct and indirect drivers at the global level – is strongly related to organismal traits in a wide range of taxonomic groups. The traits that are most likely to be lost from assemblages through extinction differ somewhat among groups, but commonly include habitat and dietary specialism, slow reproductive rate, and large body size (Owens & Bennett 2000, Lee & Jetz 2011, Cardillo et al. 2005, Fritz et al. 2009, Davidson et al. 2009, Böhm et al. 2016, Cooper et al. 2008, Dulvy et al. 2014, Bland et al. 2017, Mankga & Yessoufou 2017).

Box 2.5. Rapid evolution

Evolution is typically assumed to be a very slow process, with many species exhibiting remarkable stability over millions of years. This stability is mostly a function of precise adaptation to relatively stable environments; hence, when environments change rapidly, we might expect rapid evolutionary responses. Human actions mean that many species are facing radical changes in their environments, setting up the conditions for many populations to show rapid trait change. Figure 2.17, based on an extensive review of over 4000 rates of trait change from over 350 studies, reveals that each of the main direct drivers can provoke rapid trait change, as can natural disturbances.
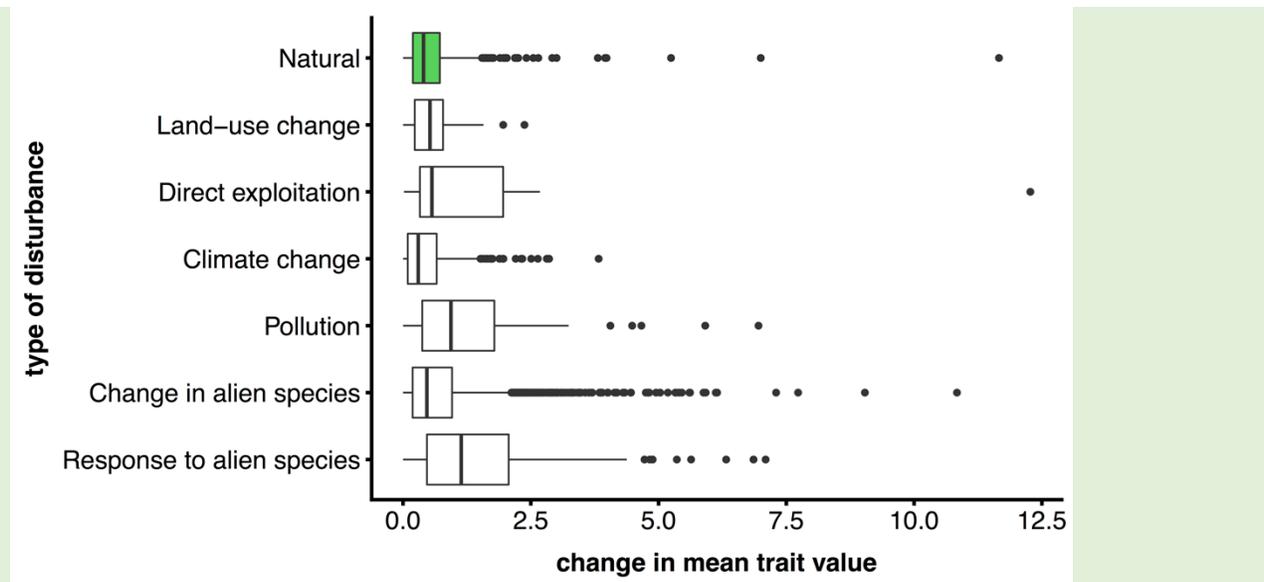
Figure 2.17 Meta-analysis of published estimates of rapid changes in trait means (expressed as the population's change in the mean trait value divided by its standard deviation) within populations that faced natural disturbances or the direct anthropogenic drivers of change. Vertical lines indicate medians, and boxes span 25th-75th centiles. Sample sizes: natural disturbance, 574 effects (49 studies); land-use change, 122 (19); direct exploitation, 18 (7); climate change, 327 (197); pollution 68 (12); change in alien species, 3329 (87); change in native responding to alien species, 223 (10).

Attributing rapid trait changes to evolution (genetic change), plasticity (direct environmental influences on individual development or behaviour), or a combination of both, takes additional focused investigation. Nonetheless, numerous case studies are demonstrating rapid evolution in response to each of the main direct drivers. For example:

• Land-use change caused significant genetic differentiation among plant populations in grassland sites facing different land uses and intensities, in all eight species tested (Völler et al. 2017)

• Direct exploitation is likely to cause evolutionary change whenever the phenotypes it targets are under genetic control. For instance, trophy hunting of bighorn sheep drives the rapid evolution of smaller horn size (Pigeon et al. 2016); while commercial fishing drives the rapid evolution of smaller size and earlier maturity (Sharpe and Hendry 2009) - although it can be hard to prove a genetic basis underlying the change.

• Climate change is driving rapid evolution in many populations and species (Merila and Hendry 2014). For instance, pitcher plant mostquitoes (*Wyeomyia smithii*) have evolved earlier pupation timing in accordance with earlier spring warming (Bradshaw and Holzapfel 2001).

• Pollution can rapidly drive evolution of tolerance (Hamilton et al. 2017), with a recent example being killifish (*Fundulus heteroclitus*) adapting to PCBs in estuaries along the eastern coast of North America (Reid et al. 2016).

Cities present novel and in many ways extreme environments and are driving rapid evolution in many species (Johnson and Munshii-South 2017, Alberti et al. 2017). Two clear recent examples are the evolution of freeze-tolerance of white clover, *Trifolium repens* (Thompson et al. 2016), and the evolution of significantly reduced dispersal another plant species, *Crepis sancta*, within 12 generations in response to urban habitat fragmentation (Cheptou et al. 2017).

Evolutionary change in these traits likely influences the ability of organisms to persist and thrive in altered environments, a phenomenon called "evolutionary rescue" (Carlson et al. 2014). Yet evolution won't always save populations or species – the outcome depends on many factors, including the demographic cost imposed by the disturbance, the strength of selection, and the genetic variation available for evolution. Hence, policy decisions that seek to maintain populations and species can manipulate these factors to maximize population persistence and productivity, and nature's contributions to people. For example, alternative harvesting regimes can drive different evolutionary changes that can have different effects on sustainability and productivity (Jorgensen et al. 2009; Dunlop et al. 2018); tailoring hunting or fishing regulations, such as maximum or minimum allowable sizes, can reduce the evolution of smaller body size and earlier reproduction (Dunlop et al. 2009). As another example, moving individuals with beneficial genotypes between populations can facilitate rapid adaptation to new climate conditions (i.e., assisted gene flow: Aitken and Whitlock 2013, McLachlan et al. 2007).

Policy decisions that influence rapid evolution can also be used to reduce the impact of harmful species, such as pest or pathogens. For instance, the rapid evolution of antibiotic resistance in many bacterial pathogens, and the rapid evolution of pesticide- and GMO-crop resistance in many crop pests, have been identified as major threats to human wellbeing (World Economic Forum 2018, Carroll et al. 2014). Hence, evolutionarily-informed policies have been used to slow the evolution of resistance (Tabashnik et al. 2008, Carroll et al. 2014); e.g., "refuges" – areas not planted with GMO crops or not sprayed with insecticides – are routinely used to prevent the evolution of resistance by insect pests to GMO crops or insecticides (Tabashnik et al. 2008, Carrière et al. 2010). Similarly, control of mosquitoes has been severely hampered by their evolution of pesticide-resistance – leading to the development of control strategies that are evolution-resistant (Read et al. 2009) or that also make use of evolution: for instance, 'gene drive' can cause the rapid evolution of phenotypes that have much lower (rather than higher) fitness, and thus may disrupt mosquito reproduction or malarial transmission (Eckhoff et al. 2016).

The widespread trait-mediated effects of drivers have caused dramatic shifts in organismal trait distributions (means and variances), though few global indicators are yet available (Figure 2.18). The *Marine Trophic Index*, which reflects the average trophic level of fish caught within multiple regions, has fallen from around 4.0 to around 3.6 in the last 60 years, because fishing

preferentially removes larger, more predatory fish (Pauly et al. 1998): the proportion of global fish biomass that is made up of predatory fish has declined by a factor of around 10 since 1880 (Christensen et al. 2014). The declining size of harvested individuals can reduce fishery productivity (Dunlop et al. 2015). On land, the median *mammalian body mass* of species within 1° grid cells has fallen by 18% (Figure 2.19; Santini et al. 2017), while a general ecosystem model (Harfoot et al. 2014) estimates that *functional richness* within 0.5° grid cells is falling worldwide.
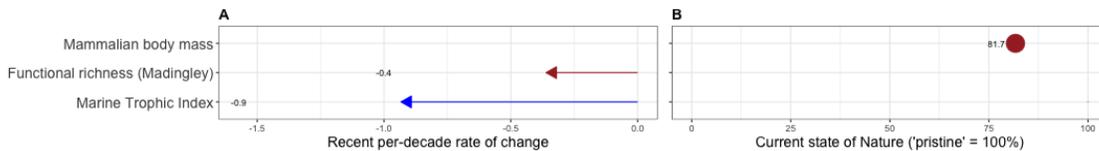

Figure 2.18: Global indicators of species traits. Marine indicators are in blue, terrestrial in brown. **A**: Trends, shown as the average per-decade rate of change since 1970 (or since the earliest post-1970 year for which data are available), ordered by rate of change. **B**: Estimated current status relative to a pristine or at least largely pre-industrial baseline. No indicators provide both trend and status. See Supplementary Material for Chapter 2 Nature for detailed information on each indicator.
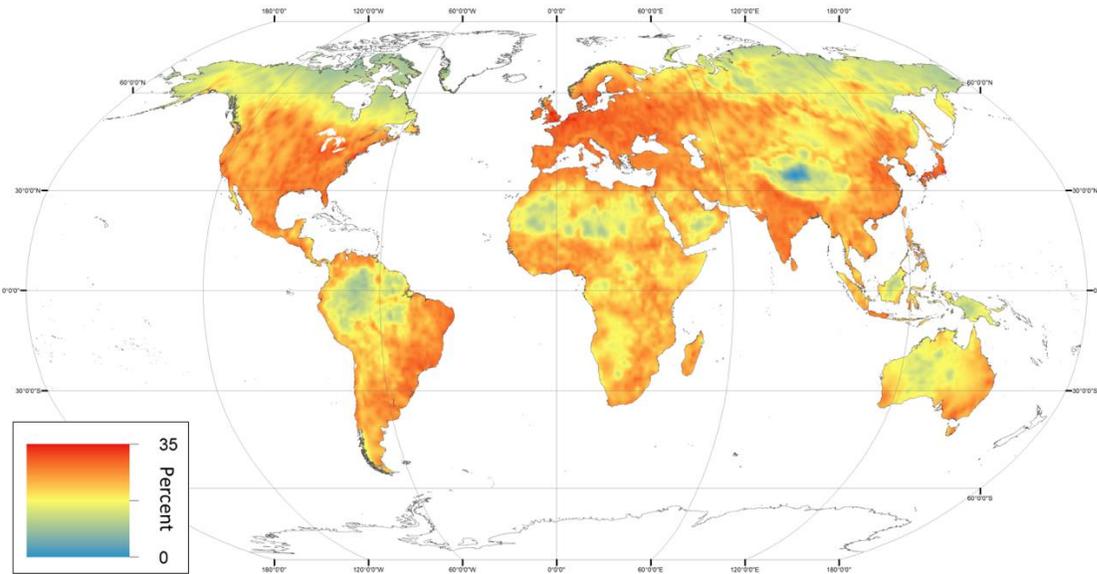

Figure 2.19: Geographic variation in the percentage reduction in median mammalian body mass as a result of species range loss caused by human impacts. Data from Santini et al. (2017).

Changes in trait means can have important consequences for population dynamics, community structure, ecosystem functioning, and – more generally – nature's contributions to people. For example, the widespread declines of large species are already profound affecting many ecosystem functions at sea and on land (Dirzo et al. 2014; McCauley et al. 2015; Ripple et al.

2014**).** Extinct terrestrial megafauna maintained a degree of openness in forest structure, giving landscapes high habitat diversity; their loss has led to more forest canopy closure and has also changed fire regimes (Johnson 2009), greatly reduced long-distance dispersal of many fruits (Pires et al. 2018) and dispersal of productivity-limiting nutrients (Doughty et al. 2013), as well as affecting many other ecosystem processes (Ripple et al. 2015). Likewise, the historical and ongoing loss of large species from oceans has reduced connectivity among ecosystems and reduced their temporal stability (McCauley et al. 2015).

Changes in trait diversity are important as well as changes in mean values, because the assemblage-level diversity in how populations respond to drivers of change underpins ecosystem stability and resilience under drivers of change (Diaz & Cabido 2001, Elmqvist et al. 2003). For instance, both among- and within-population diversity in adaptive life history traits in salmon tend to stabilize temporal variation in overall abundance and hence harvest (Schindler et al. 2014). Similarly, different plant genotypes have different effects on arthropod communities, soil microbial communities, decomposition rates, nutrient cycling, and nitrogen mineralization (Bailey et al. 2009).

Knowledge gaps: Few global indicators synthesise changes over time in organismal traits across large numbers of species, and none that does so for trait-based estimates of functional diversity, despite its ecological importance.

2.2.5.2.6 Genetic composition

Within-population genetic diversity has been lost at the rate of about 1% per decade since the mid-19th century, according to the only global meta-analysis (76 studies of 69 species: Leigh et al. 2018). Island populations in the survey tended to have lost more genetic diversity than mainland populations: those from Mauritius and the Seychelles have lost an average of 49% of their genetic diversity (Leigh et al. 2018). Support for a general human-caused decline comes from a map showing that within-species genetic diversity of amphibians and mammals tends to be lower in areas with greater human influences, especially urban areas, other settlements, and croplands (Miraldo et al. 2016; see Figure 2.2F in Section 2.2.3.1). A synthesis comparing genetic diversity estimates from wild populations facing different direct drivers found that populations whose habitat had been fragmented by land-use change have around 17% less genetic diversity than undisturbed populations (DiBattista 2008); that study found no effect of direct exploitation on genetic diversity, but another meta-analysis reported that populations of fish species that have been overfished in the last 50 years had significantly lower genetic diversity than populations of closely related species (Pinsky & Palumbi 2014). The declines in range size, numbers of populations, and population sizes of many species (Section 2.2.5.2.4) will all tend to reduce their genetic diversity (Frankham 1996).

Many farmed and domesticated plants and animals have lost genetic diversity through the extinction of races and varieties. By 2016, 559 of the 6190 domesticated breeds of mammal were recorded as extinct (including 182 breeds of cattle, 160 of sheep and 108 of pig), as well as 84 of the 2632 domesticated breeds of bird (including 62 chicken breeds and 15 breeds of duck) (FAO

2016d). A further 1500 breeds (999 mammals and 501 birds) are currently threatened with extinction (FAO 2016d). These numbers are sure to be underestimates as the conservation status of 58% of breeds remains unknown (FAO 2016d). Modernization of agriculture has sharply reduced both the numbers of crop species and numbers of varieties of those species that are cultivated (Esquinas-Alcazar 2005).

Losses of genetic variation can be permanent, or nearly so, because the forces that deplete variation (extinction, small population size, inbreeding, natural selection) typically work much more quickly than do the forces replenishing variation (speciation, mutation, recombination, gene flow). For example, the cheetah (*Acinonyx jubatus*) still shows genetic evidence of a population bottleneck around 12,000 years ago, around the same time that many other large mammals were extirpated from the area (Dobrynin et al. 2015). Similarly, hunting and land-use change have extirpated many genetically unique populations of the black rhinceros (*Diceros bicornis*), with the loss of over two-thirds of its historical mitochondrial genetic variation (Moodley et al. 2017); and the fur seal (*Arctocephalus gazella*) still has little among-population genetic variation after the commercial sealing in the 18th and 19th centuries caused populations to crash (Wynen et al. 2000).

Direct drivers have commonly been shown to reduce phylogenetic diversity (PD: Faith 1992), a measure of genetic diversity among species. In the Brazilian Caatinga, plant communities in sites that have undergone more disturbance (e.g., selective logging, fuelwood extraction and grazing) have lower PD than communities in less disturbed sites (Ribeiro et al. 2016). Costa Rican bird communities living on intensively farmed land have 900 million years less PD than those in natural forest, and 600 million years less than those on diversified agricultural land (Frishkoff et al. 2014). Worldwide, bird assemblages in highly urbanised habitats average 450 million years less PD than those in natural habitats nearby, mainly because of local extinctions (Sol et al. 2017). In some contexts, gains in PD from alien species has outweighed the PD losses from local extinctions, as in Pacific Oceanic island assemblages of flowering plants (Carvallo & Castro 2017).

Knowledge gaps: Global synthesis of patterns and trends in genetic composition is still at an early stage, with analyses so far having limited taxonomic or geographic coverage.

2.2.5.3 Status and trends of nature in land and sea managed and/or held by Indigenous Peoples and Local Communities

2.2.5.3.1 Status and trends of nature as assessed by science

(N.B. Italics denote indicators that are plotted, for Indigenous lands and for the world as a whole, in Figures 2.8 or 2.13.)

Indigenous lands have ecosystems that are more structurally intact, and ecological communities that are more compositionally intact, than the global average for terrestrial regions; and their intactness is declining more slowly. Around half of the Indigenous land mapped by Garnett et al. (2018) is still *primary vegetation*, compared with a global average of only 39% (Hurtt et al. 2018); only 7% is cultivated or urban (global average = 24%) (ESA 2017); and two thirds is

classed as 'natural' (Human Footprint score < 4), compared with only 44% of other lands (Garnett et al. 2018). The *Biodiversity Intactness Index* (BII) (Hill et al. 2018) averages 85% on Indigenous lands (vs 79% globally); a more fine-grained of BII estimate for tropical and subtropical forest biomes (*tropical forest BII*: De Palma et al. 2018) gives a lower estimate for average BII in Indigenous Lands (68%), but still higher than the global average for these biomes (62%); and *Mean Species Abundance* averages 85.5% in Indigenous Lands (vs 76.1% globally). These indicators also tend to be declining markedly more slowly in Indigenous lands than across the globe as a whole (at 33% of the global rate for the loss of *land that is not cultivated or urban*, and 68% of the global rate of loss of *tropical forest BII*).
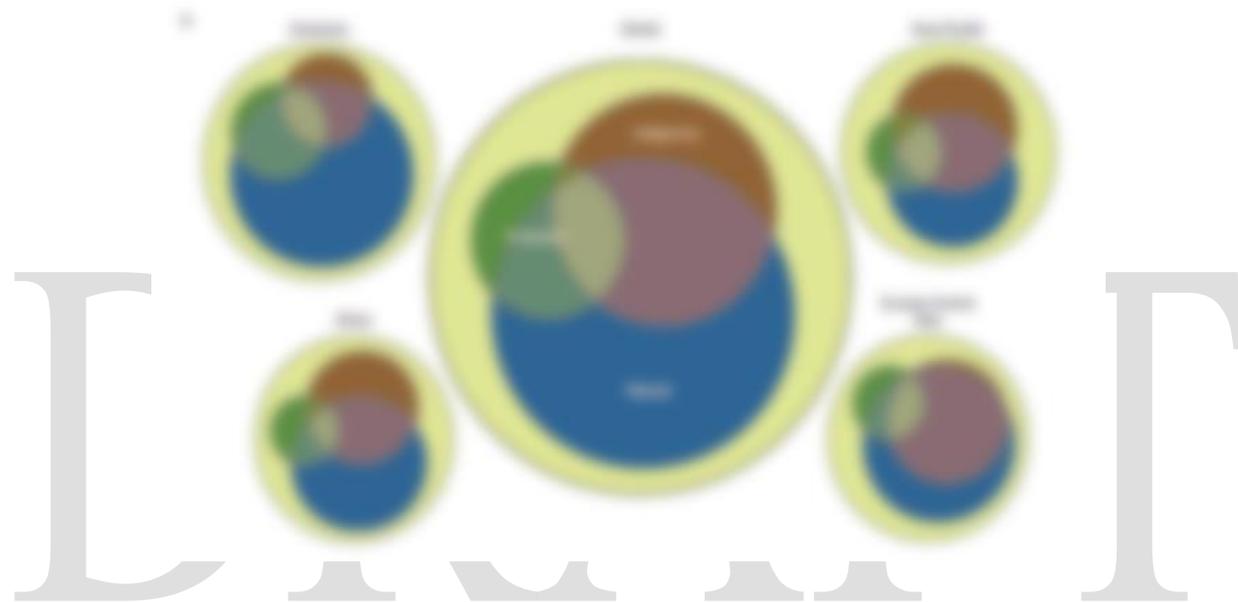

Fig. 2.20. Intersections among Indigenous lands, protected areas and natural (Human Footprint score < 4) landscapes (see following sections) globally and for each IPBES region. Circles and intersections are all proportional to area with the largest circle scaled to the land area of the Earth (excluding Antarctica) (Garnett et al. 2018).

Many of the worlds' healthiest ecosystems, and a significant proportion (and in many regions the majority) of natural land outside protected areas, are within IPLC lands (Porter-Bolland et al. 2012; Garnett et al. 2018). Several studies indicate that IPLCs reduce deforestation rates (e.g. Porter-Bolland et al. 2012, Genin et al 2013). However, to date there is not enough evidence for the conservation advantages of community-based forest management, and more quantitative case studies are needed to demonstrate causal relationships (Bowler et al. 2010, Rasolofoson et al. 2015).

No global analysis of agrodiversity trends on IPLC lands is yet available, but some biodiversity-rich lands (e.g., under shifting cultivation) have been converted to large-scale industrial food and biofuel production (Heinimann et al. 2017); and global trade increases the land area under cash-crop cultivation, decreasing local crop diversity, and pushing people to deforest, make a living on marginal areas or overexploit local biodiversity (Wolff et al. 2017). Nonetheless, lands

managed and held by IPLCs have often kept - despite agricultural modernization - a high diversity of genetic resources such as adaptive varieties and breeds (Jarvis et al 2008).

2.2.5.3.2 Trends of nature as observed by Indigenous Peoples and Local Communities (IPLCs)

IPLCs often monitor changes not only of their key natural resources but also of other salient features of nature at the population, ecosystem and landscape levels, giving them a deep understanding of multi-decadal trends in nature (Sterling et al. 2017). For example, IPLCs will often closely monitor introduced species that significantly affect natural resources important for them (e.g., Aigo & Ladio 2016, Lyver et al. 2017, Periago et al. 2017), often before they become sufficiently widespread to attract the attention of natural scientists. Culturally, ecologically or morphologically salient (cf. Hunn 1990) species are often monitored closely as well (Giglio et al. 2015, Fernández-Llamazares et al. 2016 Lykke 2000). Pastoralists frequently mention trends of populations of palatable or unpalatable species; e.g. in Europe: Fernández-Giménez & Estaque 2012; Molnár 2017; in Asia: Bruegger et al. 2014, Kakinuma et al. 2014; Hopping et al. 2016, Behmanesh et al. 2016; in Africa: Oba & Kotile 2001, Angassa & Beyene 2003, Oba & Kaitira 2006, Admasu et al. 2010; Assefa & Hans-Rudolf 2015). Ecological indicators developed and used by IPLCs are often biocultural, having both social and cultural dimensions (Sterling et al. 2017). Some of these indicators are compatible with indicators used by scientists such as those related to species composition, vegetation structure and phenological traits (cf. Harmsworth et al. 2011, Danielsen et al. 2014, Nursey-Bray and Arabana Aboriginal Corporation 2015). Other indicators – typically those with deeper social and cultural meaning – are less compatible. The selection of elements of nature monitored by IPLCs may be influenced by conservation and national policies (TEBTEBBA, 2008).

Box 2.6: Indicators of nature used by Indigenous Peoples and Local Communities

Unlike many scientific indicators that try to maximize broad comparability and therefore try not to be influenced by local context, IPLC indicators are often more closely linked to human-nature relations (Sterling et al. 2017) and are holistic in nature (Posey 1999, Berkes 2012, Inuit Circumpolar Council-Alaska 2015). Many IPLC indicators are locally tested, are intended to be locally relevant (TEBTEBBA 2008), and go back for decades (Huntington et al. 2005, Turner & Clifton 2009, Mantyka-Pringle et al. 2017). IPLCs, with a longer baseline of personal experience with the environment, may be more aware of shifts in nature (cf. changes in the Arctic, the bias in monitoring protected area management effectiveness, Corrigan et al. 2018). Some cultural memories go back hundreds or even thousands of years (Nunn and Reid 2016). Furthermore, local observations may cover many less studied, remote habitats and regions that often present environmental or technical inconveniences for scientists (Huntington et al. 2005, Fienup-Riordan et al. 2013). Finally, local monitoring systems are often independent from formal projects and financial limitations.

However, IPLCs monitoring data also have drawbacks for regional and global assessments. Notably, they are often non-quantitative and follow a fuzzy logic (Berkes and Berkes 2009,

Reyes-Garcia et al. 2016) so are less compatible with scientific monitoring protocols. Data on local trends are scattered among thousands of Indigenous and local communities, and the diverse sets of locally adapted indicators are even more difficult to synthesize globally than scientific data. IPLCs and scientific data, however, may often efficiently complement each other in helping to understand local impacts of global changes (Huntington et al. 2005, Turner & Clifton 2009, Reyes-Garcia et al. 2016).

A more detailed global synthesis of IPLC-observed trends in nature is hindered by the inherent challenges in this process, such as obtaining properly acquired Free Prior Informed Consent, the time required for adhering to local community protocols, and the lack of centralized institutions for hosting, aggregating and analyzing data of IPLCs in culturally appropriate ways.

Of the approximately 470 indicators and related 321 trend records reported in the reviewed literature, 72 % showed negative trends (Figure 2.21). Many of these (e.g. negative trends of species populations – 27.6%, negatively perceived trends regarding species composition change – 9.5%) are connected directly or indirectly to changes in nature's contributions to people that make living from nature more difficult for IPLCs (Figure 2.21B). The indicators are distributed unevenly among the Units of Analysis, but over half the trends are negative except in tundra habitats Figure 2.21A).

Figure 2.21. Opinions on trends in nature as assessed by Indigenous Peoples and Local Communities using their own indicators, split into (**A**) main ecosystem types (mostly Units of Analysis), and (**B**) per Essential Biodiversity Variable class. The analysis is based on 321 trend records published in 54 publications found in a systematic review (see Supplementary Material for Chapter 2 Nature). No data was found on genetic diversity of wild species. Opinions, whether the trends impact IPLCs in a positive, neutral or negative way, were based on local understandings of the trend, but only if it was explicitly documented in the sources. No data: no opinions on trends were provided by the publication. (Source: IPBES)

The main global trends were as follows:

- Resource availability is generally decreasing, whereas time needed or distance travelled to harvest resources is increasing (e.g. Posey 1999, Lyver et al. 2017), especially in boreal forest and tundra habitats where distribution and abundance of salient game species is changing due to climate change (Fienup-Riordan et al. 2013, Naves et al. 2015, Huntington et al. 2016).
- Declines or increases in wild species populations are among the most common indicators in almost every Unit of Analysis (26.6%, but 32.8% if indicators about their accessibility is also included), with culturally salient species often showing negative population trends (mainly plants, mammals, birds, fishes and insects, e.g., Bruegger et al. 2014, Aswani et al. 2015, Cuerrier et al. 2015, Reis-Filho et al. 2016, Reyes-Garcia et al. 2016);
- IPLCs have observed many native newcomer species arriving to their area as climate changes (e.g. southern species to boreal/arctic areas), but also the arrival and spread of new pests and aggressive alien species (e.g. Cuerrier et al. 2015, Aigo & Ladio 2016, Jandreau & Berkes 2016, Lyver et al. 2017);
- IPLC indicators recognise an increase in natural habitat loss, especially forests and grazing lands (e.g. Calvo-Iglesias et al. 2006, Ancrenaz et al. 2007, Turner & Clifton 2009, Admasu et al. 2010, Jandreau & Berkes 2016, Kimiti et al. 2016), while remnant ecosystems appear to be degrading and their biomass production decreasing (e.g., opening up of forest canopy; less biomass, more annuals and shrubs on pastures;

proportion of unpalatable plants on rangelands) (e.g. Angassa & Beyene 2003, Admasu et al. 2010, Bruegger et al. 2014, Assefa & Hans-Rudolf 2015, Behmanes et al. 2016, Jandreau & Berkes 2016);

- IPLCs have observed that the condition of wild animals appears to be deteriorating and their sizes decreasing (e.g. Moller et al. 2004, Parlee et al. 2014, Giglio et al. 2015, Naves et al. 2015, Huntington et al. 2016, Wong & Murphy 2016).

2.2.6 Global-scale analysis of attribution of trends to drivers

2.2.6.1 Challenges of synthesis

This section focuses on attributing temporal changes in the state of nature to the set of direct drivers described in sections 2.1.13-2.1.17 in Chapter 2.1, and the findings presented below are based on two extensive systematic reviews. The first (see Supplementary Material for Chapter 2 Nature for methodology) is a synthesis of natural science studies that have assessed and compared the impacts of at least two direct drivers on indicators reflecting the state of nature. This synthesis examined nearly 4000 studies and databases identified as potentially relevant, retaining 163 priority non-redundant sources (listed in Supplementary Material for Chapter 2 Nature); priority was given to large-scale studies (preferably global, but also continental or regional ones), but local studies were used when no large-scale studies were available. The second synthesis (see Appendix CC for methodology) examined how IPLCs attribute trends in nature to direct drivers. This examined 6,136 studies, retaining 192 for analysis (see details in Supplementary Material for Chapter 2 Nature). Studies were excluded from this IPLC-focused synthesis if they focused only on science-based indicators or considered community-based monitoring programmes without using locally developed indicators. The two syntheses therefore use extensive but complementary evidence bases.

Synthesising the attribution of changes in the state of nature to direct drivers is not straightforward. The complexity and high dimensionality of nature (section 2.2.3) mean that many indicators are needed to capture trends (sections 2.2.3 and 2.2.5); but indicators can differ in their metrics, sampling methods, spatial and temporal scales and resolutions, taxonomic groups, realms and regions (section 2.2.5). These syntheses therefore organise indicators using the same Essential Biodiversity Variable (EBV: Pereira et al. 2013) framework as used in sections 2.2.3 and 2.2.5, aggregating information across multiple indicators within each EBV class for robustness and generality. Specific patterns are reported for some indicators having sufficient reliable information.

There are a range of ways of comparing the importance of different drivers. For example, prevalence-based attribution can be used with IUCN Red List assessments, estimating the commonness of each driver among the listed threats (e.g. Salafsky et al. 2008; Vie et al. 2009). By contrast, Mean Species Abundance (MSA)(Alkemade et al. 2009) lends itself to effect-based attribution, because it is estimated by combining independent driver-specific dose-response models with global data on driver pressure intensity. These two approaches are in principle not directly comparable because, e.g., a driver could affect all of a set of species without being the

strongest threat to any. In order to include as wide a range of studies as possible, these syntheses have assumed prevalence-based attribution to be a reasonable approximation of effect-based attribution.

Another challenge is that studies often use threat classifications that differ from each other and from the one used in this assessment. As far as possible, threats reported in the literature were allocated to one of the five major direct drivers used in this assessment (Chapter 2.1 sections 2.1.13-2.1.17); an additional category, 'Other', was used for threats that do not fit clearly into these categories, such as fire or direct human disturbances due to recreational activities. Many studies ranked the importance of drivers instead of assessing their importance in terms of relative magnitude. Provided that the threat classification system is a good match to the one used here, this qualitative information was used and converted into quantitative estimates using a systematic approach (Hosonuma et al. 2012; see details in Appendix AA).

Although IPLCs usually possess a deep understanding of the impact of direct drivers on nature due to their closeness and direct dependence on nature for their livelihoods (Reyes-García et al. 2014; Luz et al. 2017), combining IPLC-observed driver information with natural science data presents additional problems. IPLC attribution is typically less quantitative, more scattered (geographically and thematically), and harder to aggregate globally; but provides unique insight into how drivers affect aspects of nature directly related to local livelihoods.

Section 2.2.6.2 presents the relative impacts of the different direct drivers on changes in different aspects of nature at the global level, for each of the four IPBES regions (Americas, Europe and Central Asia, Africa and Asia and the Pacific) and for each of the three global biogeographic realms (i.e. terrestrial, freshwater and marine), based on natural science indicators (Figure 2.23). Many of the attributions of global changes to drivers that are synthesised here cannot readily be partitioned into the Units of Analysis, we do not attempt to estimate the relative impacts of the different drivers at this level of resolution; the Unit of Analysis accounts in Section 2.2.7 include attributions of selected changes to drivers. Section 2.2.6.3 then synthesizes the perceptions of IPLCs about the drivers behind changes in local IPLC indicators within different types of ecosystems.

2.2.6.2   Attribution of natural science indicator trends to direct drivers

Land/sea use change is the most important direct anthropogenic driver of change in the global state of nature, with a relative impact of 30%, followed by direct exploitation (23%), climate change (14%), pollution (14%) and invasive alien species (11%) (Figure 2.23a). Threats not clearly aligned to any of these five main drivers (e.g. fire, human disturbance, recreational activities, and tourism) account for the remaining 9%.

The relative global importance of drivers varies considerably among the five EBV classes where robust comparisons could be made (too few studies assessed the relative impact of drivers of change in *Genetic composition* for comparisons to be robust) (Figure 2.23a). Land/sea use

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change is the most important driver of change for three of the five remaining EBV classes and is particularly important for *Species populations* (31.5%). Pollution is very slightly more important than land/sea use change (22.5% vs 22%) in driving changes in *Ecosystem structure* but is not in the top two drivers for other EBV classes. Direct exploitation is the most important driver of changes in *Species traits* (23.5%), with climate change second (21%). Climate change is also second for *Community composition* and *Ecosystem function*.

(a) Relative impact of direct drivers at the global scale
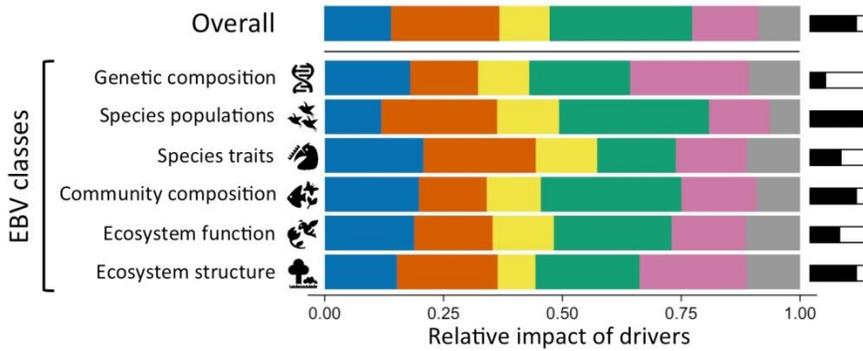
(b) Relative impact of direct drivers at the regional scale
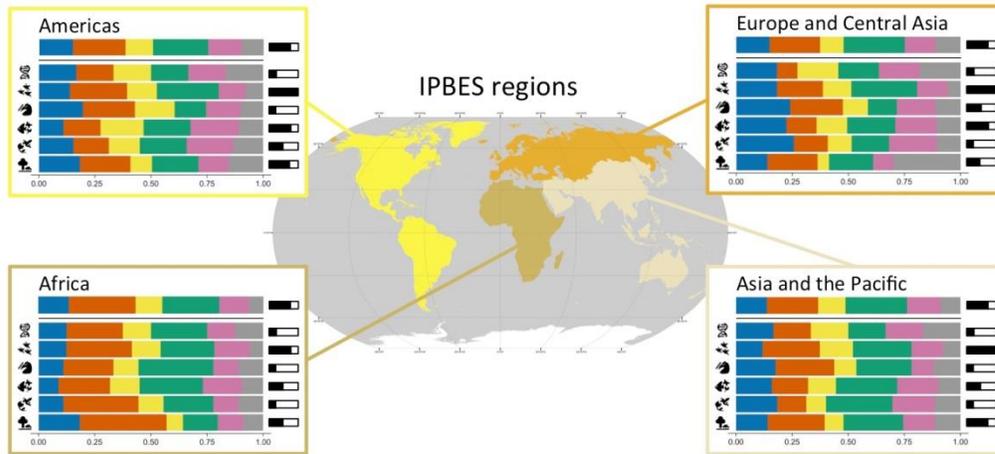
(c) Relative impact of direct drivers within realms
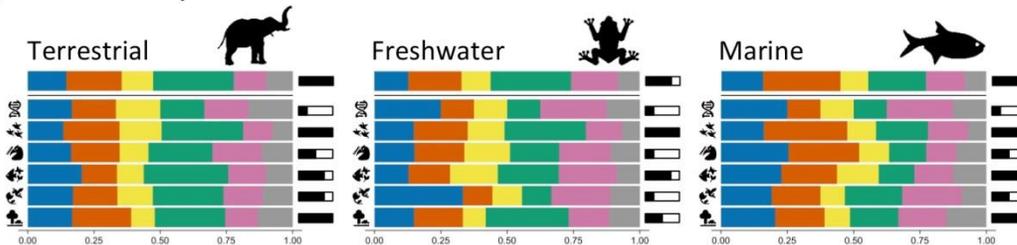
Figure 2.23: Relative impact of direct anthropogenic drivers (colour bars) on the state of nature at the global scale (a), within each IPBES region (b) and for terrestrial, freshwater and marine realms (c). The top row in each panel shows the overall pattern including all the indicators used in the analysis. The next rows show the patterns for each of the six classes of Essential Biodiversity Variables (EBV), each represented by several indicators. The width of each colour bar indicates the estimated relative importance of each driver in changing the state of nature but should not be interpreted as an absolute magnitude of the impact of each driver because both qualitative and quantitative information was combined in the analysis (see details in the main text). The degree of confidence shown alongside each row (more black = more confidence) reflects the quantity and quality of information available in the literature to estimate the relative impact of different drivers at the corresponding level of analysis (see confidence framework in Chapter 1). Note that the top row in each panel is not a simple average across the different EBV classes: some classes include more indicators and/or more studies than other classes (see degree of confidence) so have a higher weight in the estimations. A full list of studies synthesised in this figure is provided in Appendix BB, and the methodology is described fully in Appendix AA. Credits for icons: EBV classes icons created by Cesar Gutiérrez of the Humboldt Institute – Bogotá, Colombia- for GEO BON; icons for realms provided by WWF.

The relative global importance of direct drivers also varies among indicators within EBV classes, as shown in Figure 2.24 for a set of specific indicators for which sufficient information was available. Further discussion on these indicators is presented in Appendix DD.
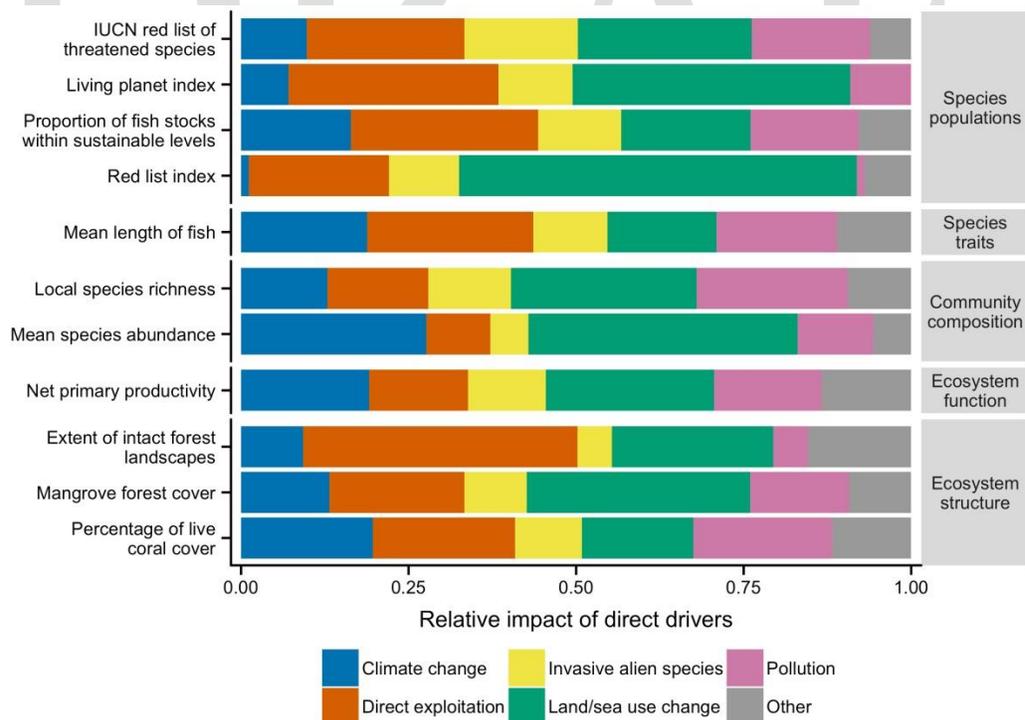

Figure 2.24 Relative impact of direct anthropogenic biophysical drivers (colour bars) on selected indicators of the state of nature for which sufficient representative information was available. Indicators are grouped according to the Essential Biodiversity Variable (EBV) framework (see

right-hand side), except that no indicators were available for the EBV class *Genetic composition*. The driver category "Other" includes threats that do not clearly belong to any of the five main drivers (e.g. fire, human disturbance, recreational activities, and tourism). The width of each colour bar indicates each driver's estimated relative importance in changing the state of nature (see details in the main text and in Figure 2.23). Further discussion on these individual indicators is included in Supplementary Material for Chapter 2 Nature.

The four IPBES regions largely reflect the global pattern (Figure 2.23b), but there are some regional differences. In Africa, the impact of direct exploitation (30%) exceeds that of land/sea use change (25.5%). In the Americas, these two drivers have a similar impact (23.5 and 25%, respectively). In the other two regions, land/sea use change is the most important driver of change in the state of nature.

Each IPBES region shows considerable variation among EBV classes. For example, direct exploitation has the highest impact on *Ecosystem structure* in Africa, whereas other threats (i.e. fires) are particularly important in Europe and Central Asia (Figure GATD-b). Although climate change is not the dominant driver across EBV classes in any of the IPBES regions, it has a particularly high impact on *Species traits*, *Community composition,* and *Ecosystem function* in Europe and Central Asia.

Land/sea use change is the most important driver of changes in the terrestrial and freshwater realms (30.5% in both cases), whereas direct exploitation is the main driver in marine ecosystems (29%) (Figure 2.23c). Direct exploitation is the second most important driver in both terrestrial (21%) and freshwater (20%) ecosystems. Climate change is not amongst the two most important drivers of change in any of the realms. In freshwater environments, pollution (17.5%) is more important than climate change (13%) whereas these two drivers have a similar impact (15% and 16%, respectively) in marine systems.

Within each realm there is considerable variation among EBV classes (Figure 2.23c). In terrestrial ecosystems, the greatest impact of land/sea use change is on *Species populations* (31%) and *Community composition* (32%). In freshwater ecosystems, this driver particularly affects *Species populations* and *Ecosystem structure* (both 31%). For marine ecosystems, the highest impact of direct exploitation is on *Species populations* (31.5%). Climate change's strongest impact on land is on *Community composition* (20%); in freshwater it is on *Ecosystem function* (33%) (but with a low degree of confidence); and in the marine realm it most affects *Species traits* (25.5%). Even if their overall importance is limited in all the realms, invasive alien species are markedly impacting some aspects of biodiversity, such as *Community composition* in freshwater ecosystems (18%).

2.2.6.3   Attribution of drivers by IPLCs

The two most important drivers of changes in nature observed by IPLCs are land/sea use change and climate change (Figure 2.25). Land-use change includes mainly conversion to intensive agriculture, urbanization and discontinuation of traditional land-management practices. For example, land use change and expansion of settlements (urbanisation) are the direct drivers of

change in rangelands most often mentioned by IPLCs in a number of African regions (Admasu et al. 2010; Assefa & Hans-Rudolf 2016; Jandreau & Berkes 2016; Kimiti et al. 2016).

Discontinued traditional land management practices (abandonment) were observed as direct causes for some changes to vegetation structure (e.g. bush encroachment, reforestation) (Oba & Kotile 2001; von Glasenapp & Thornton 2011; Babai & Molnár 2014b). Climatic changes, such as droughts and the increasingly unpredictable annual distribution of rainfalls, are important observed reasons for the decreasing biomass production and changes in vegetation structure of rangelands, which often require reorganization of traditional grazing regimes (e.g. Kakinuma et al. 2008; Admasu et al. 2010; Assefa & Hans-Rudolf 2016, Duenn et al. 2017). Altered rainfall patterns, which can influence behaviour patterns of wild animals (e.g. game or migration patterns of birds) (Turner & Clifton 2009; MacDonald et al. 2013; Kimiti et al. 2016; Ingty 2017), are also seen as important drivers of change.

Deliberate or unintentional introduction of new species can also be direct drivers of changing species pools in different habitats (e.g., inland water bodies, wetlands – Aigo & Ladio 2016, terrestrial habitats – Lyver et al. 2017; Periago et al. 2017). IPLCs report invasive alien species affecting a wide range of taxonomic groups (e.g. plants, fishes, birds) (Waudby et al. 2012; Aigo & Ladio 2016; Lyver et al. 2017; Periago et al. 2017). Overexploitation was the most often reported driver for deterioration of pasture land and tropical forests at the agricultural frontiers (e.g., Aswani et al. 2015; Fernández-Llamazares et al. 2016). Local overharvesting by companies or by local fishers motivated by commercial trade were mentioned by artisanal fishers as drivers of observed decreasing fish stocks (Carr & Heyman 2012; Aswani et al. 2015; Giglio et al. 2015; Reis-Filho et al. 2016).
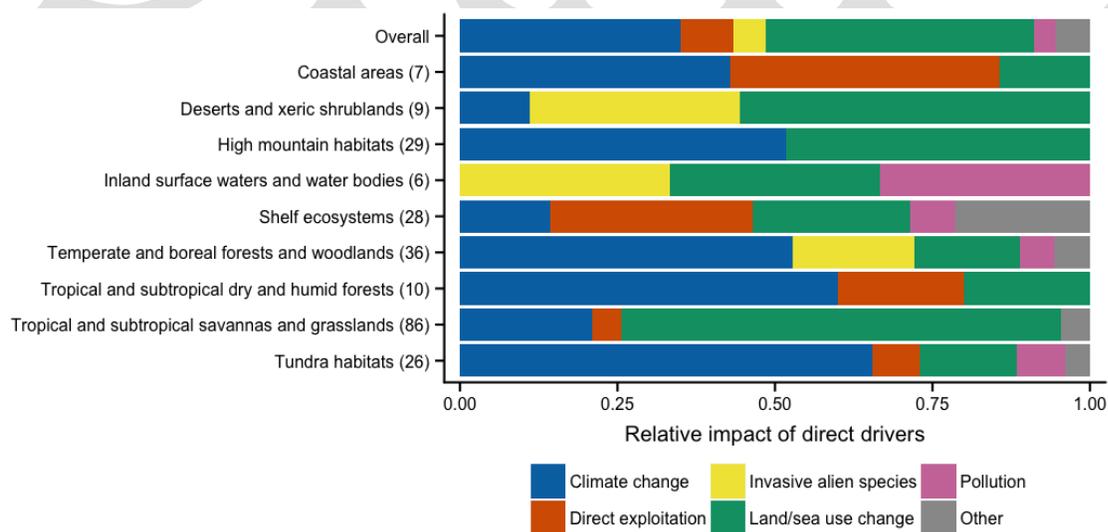

Figure 2.25 Attribution of changes in different ecosystems (rows) to direct drivers (colour bars) compiled from IPLC observations worldwide based on 470 reviewed local IPLC indicators (see the 72 publications in Supplementary Material for Chapter 2 Nature). Numbers in brackets are numbers of indicators with driver attribution. Relative impact of direct drivers is shown as their

relative frequency in the reviewed studies. The category "other" includes threats that were identified for some ecosystems during the collection of data but are not clearly linked to the five main categories (e.g. vegetation encroachment, disease and insect outbreaks).

DRAFT

2.2.7   Units of Analysis

2.2.7.1   Introduction

The Units of Analysis for the Global Assessment are a broad-based classification system at the global level, considering both the state of nature in classes equivalent to biomes, and in anthropogenically-altered biomes or 'anthromes'. The units correspond broadly to global classifications of nature and human interactions, serving the need for analysis and communication in a global policy context. The list of 17 global Units of Analysis includes 13 biomes (7 terrestrial, 2 freshwater, 3 marine and one cuts across all three) and 4 anthromes (see Chapter 1 and Figure 2.2a). All terrestrial biomes except for the cryosphere have been settled and populated by IPLCs historically, and increasingly by modern societies. The freshwater biomes reflect a simple split in relation to depth and vegetation, i.e. coarse function, and the marine biomes reflect the most basic division of oceans by depth and proximity to land. Nevertheless, the biomes reflect relatively well-known properties and variation in nature across the globe and coproduction of NCPs by people. Biomes are the lungs, heart, production center, skin and kidneys of planet earth. They cycle carbon, nitrogen and other elements; provide food and materials; process waste.

The biomes vary in state from unaltered to highly altered or degraded. The addition of anthropogenic drivers of decline to natural disturbances can impose significant cumulative impacts on biomes, with complex interactions. Some biomes remain unmodified in only a small fraction of their former range; e.g., in most regions, nearly all temperate forest has been altered or is under active human management. There is considerable variation among and within regions, and among and within biomes. Some biomes have experienced positive changes recently, as land-use practices reverse; boreal forest area has been stable for decades, while temperate forest area has expanded 10% since 1990.

Anthromes are highly altered biomes, defined by humanity's monopolization and/or maximization of one or more NCPs (i.e. distinct from degraded biomes). The main drivers of biome conversion to anthromes include large-scale commercial agriculture, local subsistence agriculture, urban expansion, construction and mining. The anthromes layer over biomes (e.g. a city in a grassland area), but some so transformed the original biome no longer exists there. Two anthromes are exclusively terrestrial, reflecting where people live and channel biological productivity to serve needs through food, timber and other types of production. In temperate biomes, conversion to anthromes and deterioration has slowed to zero or even reversed with active restoration. However, in tropical biomes, where both human population and economic growth are high, conversion rates are still high. The aquaculture and intensively developed coastline anthromes cut across terrestrial, freshwater and marine systems, and conversion of marine biomes to anthromes is at its early stages. Both aquaculture and the intensively/multiply used coastlines are likely at an early stage of acceleration (see descriptions below), and no datasets currently exist to estimate their area.

The attribution of drivers presented in this section are based on key references identified by the authors for each Unit of Analysis and is therefore different and complementary to that in section 2.2.6, which shows attributions by IPBES regions and by realms based on a global-scale systematic review of literature.

Table 2.1. Overview of some of the features on the IPBES Units of Analysis

Unit Name	Unit ID	Area (mSq Km)	NPP (gC/m2/year x10^6)	Average Relative Species Richness	Population (millions of people)	Urban Areas (Unit 9)		Cultivated Areas (Unit 10)	
						mSqKm	% of unit	mSqKm	% of unit
Tropical and subtropical dry and humid forests	1	23.49	64	0.51	2,880	0.13	0.6%	6.83	29.1%
Temperate and boreal forests and woodlands	2	32.04	69	0.17	2,003	0.49	1.5%	7.75	24.2%
Mediterranean forests, woodlands, and scrub	3	3.22	5	0.20	314	0.06	1.8%	1.58	48.9%
Arctic and mountain tundra	4	13.55	12	0.09	169	0.01	0.1%	0.70	5.1%
Tropical and subtropical grasslands	5	20.18	26	0.35	655	0.03	0.2%	4.42	21.9%
Temperate Grasslands	6	11.19	14	0.20	363	0.10	0.9%	6.27	56.0%
Deserts and xeric shrublands	7	27.89	8	0.14	788	0.06	0.2%	2.18	7.8%
Cryosphere+	11	17.71 (22.59)			-	-	0.0%	-	0.0%
Subtotals (terrestrial)		**149.28**	**198**		**7,171**	**0.88**	**0.6%**	**29.72**	**19.9%**
Wetlands	8	NA							
Inland surface waters and water bodies	13	3.65	1	0.16					

Subtotals (inland & fresh waters)		**3.65**	**1.47**				
Shelf ecosystems	14	21.16	36	0.06			
Surface Open Ocean	15		152	0.04			
		336.63					
Deep Sea	16	*		*			
Subtotals (ocean)			**187**				
		357.79					
Urban and Semiurban Areas	9	0.88					
Cultivated Areas	10	29.72					
Aquaculture Areas	12	**					
Coastal areas intensively used by humans	17	**					
Subtotals (anthromes)		**30.60**	**-**				
TOTAL			**387**		**7,171**	**0.88**	**29.72**
		510.72					

Notes

Empty cells show where numbers are not applicable

All values are in 2015, unless otherwise noted.

+ Area of terrestrial cryosphere = 17.71 mSqKm. Arctic and Southern Ocean annual sea ice extent has averaged 22.59 mSqKm for the ten years from 2008-2017.

* Same as/included in Unit 15

** Units have no calculable area. There are no databases for aquaculture locations (terrestrial, freshwater and marine) from which area can be calculated. 'Intensely and multiple used coastline' is currently undefined in terms of area, as the coastline is a linear feature. Global datasets are also not available for estimating its length or area.

2.2.7.2   Tropical and subtropical dry and humid forests

Tropical and subtropical forests cover about 52% of global forested land (FAO 2015b; Keenan et al. 2015), holding an aboveground carbon stock of 190-220 billion tons (Saatchi et al. 2011; Baccini et al. 2012; Liu et al. 2015), representing about 70% of the carbon in forests globally (Yingchun et al. 2012), and 35% of terrestrial GPP (Beer et al. 2010). These ecosystems harbor the greatest biological diversity globally, containing for example the ten hotspots with the

greatest total number of endemic higher terrestrial vertebrates (Mittermeier 2004; Mittermeier et al. 2011) and the greatest number of threatened species.

This Unit plays a vital role in local to global climate regulation, through complex hydrological and biogeochemical dynamics, mainly of CO_2 and water vapor (Bonan 2008). The Amazonian rainforest keeps the air humid for over 3,000 km inland (Salati et al. 1979), and transpires twenty billion tons of water daily (Nobre 2014).

Globally, tropical and subtropical forest area has declined from 1990-2015. All top ten countries reporting the greatest annual net loss of forest area for 2010-2015 belong to this Unit (FAO 2015b). The rate of loss of tropical forests was 10.4 M ha yr^{-1} in the 1990s, slowing to 6.4 M ha y^{-1} in 2010-2015 (Keenan et al. 2015). For subtropical forests these numbers were 0.4 M ha yr^{-1} and 0.0 M ha yr^{-1}, respectively. These averages mask high variance between regions, as well as within regions and countries, with highest losses in South America and Africa (Hansen et al. 2013). For example, while Brazil showed a reduction in annual forest loss from 2000-2012, increases were measured in all other regions.

Land use change is the main driver of forest loss in tropical and subtropical regions (Meyfroidt & Lambin 2011; Newbold et al. 2014; FAO 2016b); other subdrivers vary in importance among and within regions (DeFries et al. 2010; Boucher et al. 2011; FAO 2016b). Overall, the main cause of deforestation is large-scale commercial agriculture (e.g., cattle ranching, oil palm, soy, and cocoa) (40% of deforestation), followed by local subsistence agriculture (33%), urban expansion (10%), infrastructure (10%) and mining (7%) (Hosonuma et al. 2012; FAO 2016b). Forest degradation is driven mainly by timber and logging (58%), fuelwood/charcoal (27%), uncontrolled fires (10%), and urban expansion (5%). Recognition of IPLCs' territories helps buffer deforestation in the Amazon (Soares-Filho et al. 2010), and local farmer communities can contribute to reforestation (Jacobi et al. 2013).

Habitat loss and degradation are the main causes of reductions in species richness and abundance (Newbold et al. 2014; WWF 2016), while habitat conversion and harvesting are the main threats to Threatened plant species in tropical forests (Brummitt et al. 2015). Main trends perceived by IPLCs include the loss (or introduction) of salient large mammals (e.g. elephant, pecary) (Ancrenaz et al. 2007; Sahoo et al. 2013) and the proliferation or collapse of plant species (e.g. medicinal plants) (Fernández-Llamazares et al. 2016).

Tropical and subtropical regions are projected to experience extreme climatic conditions earlier than other regions, such as Boreal forests, Tundra and Taiga (Beaumont et al. 2011) (Beaumont et al. 2011). Extreme climate events in the last two decades (Chen et al. 2010; Satyamurty et al. 2013; Marengo et al. 2013), interacting with other factors such as deforestation and fire, have caused large-scale long-lasting impacts on forest structure and function, affecting hydrological and carbon cycles (Davidson et al. 2012; Qie et al. 2017).

Positive trends in forest cover are reported in thirteen tropical and subtropical countries containing 6.4% of global tropical and subtropical forest area (Appendix 2.2.7.2). These countries have transitioned from net forest loss to net gain, mainly driven by planted-forest expansion (Keenan et al. 2015; Sloan & Sayer 2015; FAO 2016b).

2.2.7.3 Boreal and temperate forests

Boreal and temperate forests comprise one third and a quarter of global forest cover, respectively (FAO 2015b), covering 1.91 billion ha (FAO 2015b). They experience a cold continental climate, with a growing season of <130 days (temperate) and >140 days (boreal). Boreal forests sustain a low richness of coniferous trees that withstand freezing and extended dormant periods, with two abundant deciduous genera. The temperate zone has many continental endemic deciduous species, with some common genera, such as pines. The boreal biome is primarily in Canada, Russia, and Scandinavia, while the temperate zone occurs in both hemispheres, on six continents. Highly productive temperate rainforests occur on the west coast of North America, Chile, New Zealand, and Australia.

Boreal forest area did not change between 1990 and 2015 (FAO 2015b) and 43.8% of the remaining global "Intact Forest Landscapes" are boreal (Potapov et al. 2008). Nearly two-thirds of boreal forests are currently under management, mostly for timber (Gauthier et al. 2015). Virtually all temperate forests in most regions of the world are managed; temperate China and Europe were largely deforested by the 1500s, many countries have lost > 90% of their forest cover (Kaplan et al. 2009), and there are no large intact or primary forest areas (Kishnasway and Hanson 1999). Temperate forests have increased by about 67 million ha since 1990, largely due to planting in China and farm abandonment globally (Campbell et al. 2008, Yin et al. 2005, FAO 2015b, Keenan et al. 2015), but young secondary forest is much less rich in biodiversity than primary forest. Over 350,000km^2 of intact forest landscapes (i.e., large areas of forest or natural mosaic, free from evident signs of human disturbance) were lost from temperate and boreal forests between 2000 and 2013 (Potapov et al. 2017), showing continuing deterioration in the condition of primary forest within this Unit of Analysis.

The boreal forest is the largest store of terrestrial carbon (Pan et al. 2011, Gauthier et al. 2015, Bradshaw and Warkentin 2015), over 75% of which is in soil organic matter (Rapalee et al. 1998, Bradshaw and Warkentin 2015). Boreal forest has sequestered 0.5 Pg C/yr since 1990, accounting for 20% of the annual terrestrial forest carbon sink (Pan et al. 2011, Kurz et al. 2013), but not all boreal forests are sinks owing to increased fires and respiration due to climate change (Hadden and Grelle 2017). Between 1990 and 2007 temperate forests have stored a net 0.72 Pg C/yr (Pan et al. 2011).

Both biomes are highly susceptible to climate change (Setterle et al. 2013), increasing fire risk (Bradshaw et al. 2009), in part because of low boreal productivity and high susceptibility of peat and permafrost soils. Other climate drivers include moisture stress, warmer temperatures,

increased insect infestations, N deposition, and CO2 fertilisation (Silva et al. 2010, Kint et al. 2012). Drier, warmer boreal forests will store less carbon due to moisture stress (Ma et al. 2012), becoming a net source of greenhouse gasses (Kurz et al. 2013, Flannigan et al. 2000, 2009), despite increased productivity in northern open taiga forests (Goldblum and Rigg 2010, Boucher et al. 2017). A warming climate may result in release of the huge carbon store in frozen boreal peat soils (Schaefer et al. 2011). Projections suggest shifts in forest distribution, depending on dispersal ability among tree species (e.g., Soja et al. 2007). Large areas in the boreal forests are inhabited by IPLCs in Eurasia and North-America, who report changing animal population trends (e.g. increasing moose, decreasing caribou, decreasing bird species, e.g. geese) and changing migration patterns, due to climate change (MacDonald et al. 2013, Lyver et al. 2017).

Invasive species and diseases have become a major driver of tree mortality in some temperate forests (Adams et al. 2012, Charru et al. 2010), and diseases are a developing problem in plantations (Winfield et al. 2015). Some planted trees are invasive in temperate forests, e.g. *Acacia* (Yelenik et al. 2004, Lorenzo et al. 2011). Temperate regions have high numbers of threatened and endangered species, including >500 tree species (Oldfield et al. 1998, IUCN 2017), and there have been extinctions, including passenger pigeon (*Ectopistes migratorius*). No boreal plant or animal species has gone extinct but there have been national-level extirpations.

2.2.7.4 Mediterranean forests, woodlands and scrub

Mediterranean forests, woodlands, fynbos and scrub are discontinuously spread in five continents and twenty-two countries (Dallman 1998). They cover 4 million km^2 (2% of total land area) in southern Europe and northern Africa (Mediterranean Basin), South Africa (Western Cape), northwestern America (e.g. California chaparral), southern America (Chilean matorral), and southern Australia. These regions harbour an extremely high diversity of species originating from almost all known biogeographic realms of the world including new land races (Blondel et al., 2010; de Cortes Sánchez-Mata and Tardío, 2016) and include five biodiversity hotspots of global importance (Myers et al., 2000, Mittermeier et al. 2011). Vegetation types are coniferous or (mostly evergreen) broadleaf forests and woodlands, savannahs and grasslands, scrublands and mosaic landscapes, resulting from a strong interaction between heterogeneous environmental conditions and a long-lasting influence of human activities (Blondel, 2006). The Mediterranean biome has the second lowest level of land protection among terrestrial biomes (Hoekstra et al., 2005) and is projected to experience the largest future proportional loss of biodiversity (Sala et al., 2000; Malcolm et al., 2006).

Mediterranean terrestrial ecosystems are highly sensitive to the combined effect of global change drivers and specific driving forces, including climate change, land-use transformations and fires (Templado, 2014; Valladares et al., 2014; Barredo et al. 2016). With the particular geology of Mediterranean systems, these changes have resulted in more frequent and intense fires, water scarcity, land degradation and habitat fragmentation. The unit is increasingly becoming vulnerable (Klausmeyer and Shaw, 2009; Batllori et al. 2013) and future outcomes are difficult to predict (Doblas-Miranda et al., 2015, 2017; Voltz et al., 2018). Recent shifts in fire regime

modify the composition of the vegetation (from coniferous forests to landscapes dominated by broadleaf trees, scrub and grasslands) and decrease its further resilience to fires (Gil-Tena et al., 2016), with strong impacts on key NCPs such as water supply, carbon storage and food production and a possible switch to a different kind of ecosystem. While Mediterranean forests provide various material NCPs (Bugalho et al., 2011), scrublands mostly provide non-material or regulating NCPs (e.g. pollination, reduction of extreme wildfire hazard, key habitats for biodiversity).

IPLCs have been using fire to promote herbaceous vegetation and useful game or plant species (Pechony & Shindell, 2010; Valladares et al., 2014). Such historical practices and other land-use legacies combined with more recent driving forces, such as land abandonment and fire suppression strategies, have been playing a major role in reshaping the Mediterranean landscapes (Blondel, 2006; Marlon et al., 2008; Valladares et al., 2014; Gauquelin et al., 2016).

Although Mediterranean biodiversity is facing multiple threats and is declining strongly, some driving forces may be turned into conservation opportunities. For instance, large carnivores have been recolonizing abandoned landscapes in many rural areas of the Mediterranean Basin. Although land abandonment and subsequent vegetation encroachment generate conservation concerns, this process is now also considered as an opportunity for rewilding landscapes and exploring new avenues in areas where the socioeconomic context becomes incompatible with the maintenance of traditional agricultural practices (Navarro & Pereira, 2012; Ceauşu et al., 2015).

2.2.7.5   Arctic and mountain tundra

Tundra vegetation, composed of low-growing herbaceous plants, shrubs, mosses, and lichens, grows beyond the cold limit of tree growth. Two types are recognized: mountain tundra at high elevations, and arctic tundra at high latitudes. Arctic tundra is found in Russia, Canada, the U.S., and Greenland but is not present in Scandinavia, Iceland, or the Aleutian Islands (Walker et al. 2005, CAFF 2013). This distribution corresponds roughly with the distribution of permafrost in soils, while mountain tundra soils have no permafrost. One effect of permafrost is that water from snow and rain is retained in the surface layers of soil; plants grow better in these moist soils than in the drier soils of mountain tundra. Species richness in the tundra is low; for example, the arctic tundra contains only 9% of the world's species of plants and animals.

The low numbers of people who live in the tundra regions have little effect on the native plants and animals. High plant productivity and low predator densities in arctic tundra (Bhatt et al. 2017) support many migrating animals such as reindeer/caribou, muskox, fish, and birdlife including millions of geese. Harvest of these animals supports Indigenous People and recreational hunting in temperate regions. In general, both ecosystems are still functionally intact, though in some areas used for seasonal herding, impacts are notable. Arctic and high mountain tundra are recognised as water towers (Viviroli et al 2007; Chettri et al 2012), but they are sensitive to multiple drivers including climate change (Myers-Smith et al. 2015; You et al 2017).

There are indications of higher warming in high mountains (Shrestha et al. 1999, Sun et al 2017) resulting in species range shifts (Gottfried et al. 2012; Pauli et al. 2012; Tape et al. 2016; Liang et al. 2018), phenology change (Bjorkman et al 2015; Tao et al. 2018) and low plant productivity (Bhatt et al. 2017). The arctic region is warming at roughly twice the global average (Pithan and Mauritsen 2014), resulting in a warmer, wetter, and more variable environment. The permafrost in the high arctic has warmed by more than 0.5°C since 2007-2009 (AMAP 2017); as a result, microbes release large amounts of carbon gases from the plant debris previously frozen in the soil (Schuur et al. 2008). The carbon stored in the upper few layers of arctic soil is equal to twice the carbon in the world's atmosphere (Tarnocrai et al. 2009).

These transformations have profound implications for people, resources, and ecosystems (Arctic Council 2016). IPLCs in arctic tundra report that they are already significantly challenged by changes to weather and ice conditions as well as by climate-induced shifts in hunting opportunities (e.g. fewer safe boating and hunting days, changing ice melting patterns), the animals they hunt, or the size of the grasslands they use for pastures (Parlee et al. 2014, Cuerrier et al. 2015, Huntington et al. 2016). Mountain IPLCs perceive degrading rangeland conditions because of climate change (e.g. fewer flowers, height of the vegetation, reduced quantity of forage plants, more bare soil on pastures) (Hopping et al. 2016, Ingty 2017), exacerbating alterations in mountain vegetation from high altitude pasturing for millennia (which has lowered the treeline and increased tundra in many mountain ranges) (Catalan et al. 2017).

2.2.7.6 Tropical and subtropical savannas and grasslands

Tropical savannas and grasslands cover about one fifth (~33 million km^2) of the global land surface (Scholes & Walker 1993, Ramankutty & Foley 1999, Beerling & Osborne 2006). The ecosystem services they provide sustain the livelihoods of one-fifth of the world's people, and they are also home to majority of the world's livestock and much of its charismatic wildlife (Solbrig et al. 1996, Sankaran et al. 2005, Parr et al. 2014, Lehmann et al. 2014).

Savannas and grasslands are ancient ecosystems (originating 8-10 Mya) that support unique biodiversity (Bond & Parr 2010, Ratnam et al. 2011, Veldman et al. 2015a, Murphy et al. 2016). The misconception that they are 'derived' from forests through deforestation and other land-use processes and are therefore somewhat "degraded" has resulted in mismanagement of their biodiversity, and conversion to other land uses such as agriculture and tree plantations (Bond & Parr 2010, Parr et al. 2014, Veldman et al. 2015a, Murphy et al. 2016, Ratnam et al. 2016). It is estimated that ~ 6.7 million km^2 of savanna, grassland and steppe habitats were converted to croplands between 1700 and 1992 (Ramankutty & Foley 1999), with >80% of grassland and savanna habitats being converted to anthropogenic land uses by 2000 (Ellis & Ramankutty 2008, Ellis et al. 2010). Currently, the savannas of northern Australia are the least impacted savannas (Murphy et al. 2016) while neo-tropical savannas are amongst the most threatened (Strassburg et al. 2017), globally. Very little of Asia's savanna and grassland habitats remain (Lambin et al. 2003, Miles et al. 2006, Murphy et al. 2016).

Species richness in tropical savannas and grasslands can be quite high, and in some cases comparable to forests (Murphy et al. 2016), with the Neotropics and Afrotropics especially diverse (Murphy et al. 2016). In forests much of the diversity resides in the tree layer, but grasses and forbs contribute substantially to plant species richness in tropical savannas and grasslands (Sankaran 2009, Bond & Parr 2010, Ratnam et al. 2016, Murphy et al. 2016).

Grazing and fire are integral features of savannas and grasslands and essential to their persistence (Scholes & Archer 1997, Sankaran et al. 2004, Bond 2008, Ratnam et al. 2011, Bond & Parr 2010, Parr et al. 2014). Semi-nomadic and transhumant grazing systems seem to better adapt to and cope with unpredictable climates that characterize these ecosystems than settled and paddocked animal husbandry. Local pastoralists use diverse indicators to understand pasture degradation and regeneration, such as adverse changes in woody or shrubby vegetation, or of unpalatable species (Lykke 2000, Angassa & Beyene 2003, Admasu et al. 2010, Kimiti et al. 2016, Jandreau & Berkes 2016). Active fire suppression can alter species composition and lead to establishment of forest tree species at the expense of savanna trees in more mesic areas (Bond 2008), and litter build up that fuels more intense fires when they do occur (Stott 1990, Ratnam et al. 2016). Invasions by exotic species, both grasses and trees, may have negative impacts on the native flora and fauna, and may also alter the frequency, intensity and spatial extent of fires (D'Antonio & Vitousek 1992, Rossiter et al. 2003, Hoffmann et al. 2004, Hiremath & Sundaram 2005, Aung & Koike 2015, Ratnam et al. 2016).

Carbon schemes such as REDD+ can undermine grasslands by promoting tree planting (Lehmann 2010, Parr et al. 2014, Veldman et al 2015b, Bond 2016, Ratnam et al. 2016, Abreu et al. 2017, Strassburg et al. 2017, Griffith et al. 2017). In this context, it becomes particularly critical to distinguish 'derived' from 'old-growth' grasslands and savannas, to avoid the significant costs of misguided afforestation of the latter.

Climate change will alter the tree-grass balance, in most continents leading to shrub encroachment and woody thickening (Bond & Midgley 2000, Fensham et al. 2005, Sankaran et al. 2005, Bond 2008, Good & Caylor 2011). Savanna responses to different global change drivers are likely to vary both regionally, and across continents (Higgins & Scheiter 2012, Lehmann et al. 2014), due to varied vegetation-fire-climate linkages.

2.2.7.7   Temperate grasslands

Temperate grasslands comprise steppes, prairies and pampas, as well as some high-altitude veld, forest-steppes and wood-pastures, covering an area of 13 million km² (White *et al.*, 2000; Dixon *et al.*, 2014), or 5-10% of the global terrestrial surface. Temperate grasslands have a high biodiversity of mammals and birds, and huge stocks of carbon stored in their soil. Total carbon stocks have been estimated at 450 – 550 Gt C (18 – 31% of global terrestrial carbon, White et al., 2000) with a correspondingly high potential for carbon sequestration. The capacity to store

carbon varies greatly between temperate grassland types and debate is ongoing regarding estimating this capacity (Sommer and de Pauw, 2010; Schierhorn et al., 2013; Wiesmeier et al., 2015).

Several global hotspots for vertebrates and vascular plants (Mittermeier et al, 2004) overlap with temperate grasslands. The Eurasian steppes host the largest long-distance ungulate migrations on the planet (Tucker and et al., 2018). North American prairies are relatively recently formed which is why despite massive loss of area relatively few species are at risk of extinction (Risser, 1988).

No other biome has experienced the level of degradation and conversion as temperate grasslands (Henwood, 1998; Hoekstra *et al.*, 2005). In the last century ca. 60% have been converted (White *et al.*, 2000), and <10% remain in North America and Europe, with continuing decline (Molnár *et al.*, 2012, Gauthier, 1988, Korotchenko, 2012). By contrast, <1% are converted in Mongolia. Important drivers of change in temperate grasslands are habitat conversion, fragmentation by transport infrastructure, and to a lesser extent local overgrazing. Most temperate grassland plants are adapted to grazing, yet excessive grazing or overhaying has led to degradation in many Eurasian grasslands (Wesche *et al.*, 2016) and in parts of South America (Pineiro *et al.*, 2006). Invasive species are increasingly problematic, particularly in North America and South Africa (Grace *et al.*, 2001; Morrow *et al.*, 2015; Han and Young, 2016). Decreasing productivity of temperate grasslands and changes in composition towards unpalatable species are the most frequently cited trends (Bruegger et al. 2014, Kakinuma et al. 2014). North American grasslands continue to disappear, at rates equivalent to deforestation in the Amazon, due to conversion to cropland and excessive grazing (Ceballos et al., 2010; Wright and Wimberly, 2013). Extremely rapid development threatens the integrity of Mongolia's vast steppe (Batsaikhan et al., 2014).

For traditional pastoral communities, provision of livestock forage, dung as a fuel and the open landscape are the key NCPs provided by temperate grasslands. Conversion to agriculture has slowed down and, in some regions reversed (e.g. Eurasian grasslands), with large-scale farm abandonment in e.g. Russia and Kazakhstan (Jírová et al., 2012; Chen et al., 2013). In China some restoration has commenced (Ren et al., 2016). Shifts to market economies have reduced grazing pressure of livestock in several regions, including Kazakhstan and western Russia (Kühling et al., 2016), and Patagonia (Coronato et al., 2016). Where traditional mobile pastoral practices persist, such as in Mongolia, rangelands are still relatively intact pointing to the importance of ILK and mobility for sustainable use of highly variable rangelands (Bilegsaikhan et al., 2017).

Levels of formal protection of temperate grasslands are low, at about 3.4 – 5.0% of global area (Henwood, 2012), lower than in other major terrestrial biomes. Protection is particularly low (≤ 2%) in South American pampas and the velds of Southern Africa and Australia (Peart, 2008).

2.2.7.8   Deserts and xeric shrub lands

This unit comprises large expanses of arid and hyper-arid lands in tropical and subtropical latitudes characterized by sparse often discontinuous vegetation and large expanses of bare soil. Deserts cover a total of over 33.7 million km^2, representing almost 25 per cent of the terrestrial surface of the planet (UNEP 2006). Herbivory by large and medium-sized mammals that have evolved to these dry and sparse vegetation conditions is a distinctive feature of these habitats.

Deserts and xeric habitats are characterized by severe shortage of water and are classified as arid and hyper arid with a precipitation to potential evapotranspiration (P/PET) ratio of 0.05 – 0.20 and < 0.05 (Sorensen 2007). Deserts may be hot (ground temperatures up to 80 ⁰C) or cold, mainly dependent on altitude. Both deserts and xeric shrub lands can have a dense herbaceous/grassy vegetation after the rains for very short periods of the year. The desert biome holds on average an abundance of original species of 68%, highly adaptive to severe climate conditions (UNEP 2006).

The deserts of the world occur in six biogeographical realms (UNEP 2006), with varying degrees of anthropogenic influence: Afrotropic deserts south of the Sahara in Africa and in the southern fringe of the Arabian Peninsula (2.7 million km², mean population density of 21 p/km² and a relatively high human footprint; Australasian deserts in the Australian heartland (3.6 million km², less than 1 person per km², and the lowest human footprint); Indo-Malay deserts, south of the Himalayas (0.26 million km², mean population density of 151 p/km², the most intense human use); Nearctic deserts in North America (1.7 million km², high population density of 44 p/km² due to urbanization, and the second highest human footprint); Neotropic deserts in South America (1.1 million km², a population density of 18 p/km² and a lower human footprint than in North America); and Palearctic deserts in Eurasia north of the Himalayas and in north Africa including the Sahara (63 % of all deserts, covering 16 million km²; a density of 16 p/km², and the second lowest human footprint on the planet, possibly because of inaccessibility and extreme aridity. The flat Sahara and Arab deserts contrast with the mountain deserts of Central Asia.

Deserts and their fringes are currently home to some 500 million people, about 8% of the global population. Traditionally deserts support hunter-gatherers, pastoralists and farmers (in oases and along rivers). Poverty affects many people living in deserts (UNEP, 2006). However, contrary to common belief, deserts are not a final stage of desertification but are natural ecosystems, providing many life-supporting services to mankind.

The main drivers of degradation are urbanization, tourism, intensive agriculture, mining, military operations and climate change. Biodiversity decline in deserts is expected to reach 58% of original species in 2050. Desert wilderness areas are expected to decline from 59% of total desert area in 2005 to 31% in 2050 (UNEP, 2006).

2.2.7.9 Wetlands

Wetlands are permanent or temporary, freshwater, brackish or marine areas, where water either covers the soil or is at or near its surface, either year-round or seasonally. They include

floodplains, bogs, swamps, marshes, estuaries, deltas, peatlands, potholes, vernal pools, fens and other types, depending on geography, soil, and plant life. Their global extent remains uncertain (Davidson et al 2018), but inland wetlands are estimated at 12.1 million km^2, or 6% of the world's land surface (Reis et al. 2017, Ramsar 2018). Wetlands contain about 12% of the global carbon pool, highest in peatlands (Ferrati et al. 2005, Joosten et al 2016, Ramsar 2018). Though valuation of NCP is often problematic, wetlands are estimated to contribute 21.5-30.0% of the value of global NCPs (Kingsford et al. 2016). Estuaries support around 70% of people worldwide (Halpern et al. 2012), contributing food, freshwater and protection from erosion, natural hazards and pollution (MA 2005, Costanza et al 2014, Russi et al 2013, McCartney et al 2015). They are also often culturally important to IPLCs, often in relation to intangible (e.g. sacred) values (Pyke et al 2018, Verschuuren 2006, Ramsar 2018).

Natural wetlands are declining rapidly: by 0.82-1.21% per year (Dixon et al 2016, Davidson et al 2018); by 31% between 1970 and 2008 in areas studied (Dixon et al. 2016), and by 87% between 1700 and 2000 (Davidson 2014). Historical losses were mostly inland (Davidson 2014), whereas current declines are predominantly coastal (Dixon et al. 2016). Conversely, human-altered wetlands – which make up about 12% of the global total – are increasing, especially in southern Asia and Africa, mainly through conversion of natural wetlands into paddy fields, which now cover 1.3 million km^2 (Junk et al. 2013; Davidson 2014; Ramsar 2018, Davidson et al 2018). Rice paddies deliver multiple NCPs, including pest control, soil fertility and fish production (McCartney et al 2015). Globally, IPLCs have many traditional wetland management systems. For example, the most biodiverse Norwegian swamp woodlands are managed by traditional grazing and hay mowing (Natlandsmyr and Hjelle 2016).

Changes in the water inflows and abstraction, and structural modifications (e.g. drainage and conversion) all directly drive the loss of inland wetlands (Ramsar 2018). Indirect drivers include overfishing, intensive wood harvesting (e.g. in wetland forests), peat extraction, and sand and gravel extraction for construction (Ramsar 2018). The two largest peatlands in the world (northeastern Peru and Republic of Congo) are threatened by commercial agriculture, transport infrastructure, and oil palm and timber concessions (Pearce 2017). In estuaries, increased fluvial sedimentation due to unsustainable land-use or climate change can significantly reduce fish and benthic diversity (Nicolas et al. 2010).

Freshwater marshes support disproportionately high biodiversity for their size (Kingsford et al. 2016), and several wetland types found in a mosaic with forests and mires, are important for biodiversity but poorly studied (Gupta et al. 2006; Struebig et al 2006). Wetland biodiversity is declining globally, with 25% of assessed species threatened with extinction (Ramsar 2018); 45% of mammals and 33% of birds in the South Asian Tropical Peat Swamp Forests are Near-Threatened, Vulnerable or Endangered (Posa et al. 2011). The fraction of wetland area under formal protection varies widely depending on definitions used, ranging from 11.3% to 20.4% (Reis et al. 2017).

Climate change is already a major driver of wetland structural change and influences water volumes, flows, temperature, invasive species, nutrient balance and fire regimes (Erwin 2009, Finlayson 2016). The importance of wetlands for carbon sequestration is increasingly recognized, and their loss can trigger further carbon release; annual emissions of carbon due to

peat oxidation in Indonesia are equivalent to emissions from fossil fuel burning in Canada (Pearce 2017).

Positive actions on wetlands are expanding, particularly in the USA and Europe, where wetland restoration efforts are increasing (Reis et al. 2017), including monitoring of birds (Heldbjerg et al 2015) and protection of peatlands. Numerous benefits from restoration have been documented (e.g., Erwin 2009, Reis et al. 2017); the incorporation of diverse perspectives, including indigenous and local knowledge, in wetland management is crucial for effective restoration (Russi et al 2013). However, landward migration of estuaries will depend on the availability of habitats and coastal development.

2.2.7.10 Urban/semi-urban

Urban and semi-urban areas cover approximately 88 Mha, less than 0.6% of the world's land surface (Goldewijk et al. 2017), on which 54% of the world's population lives (World Bank 2017). Urban expansion now is more rapid, more extensive and fundamentally different from how urban areas grew in the past (Seto et al. 2010). Europe and North America dominated urban growth from 1750-1950, but in 1950-2030, the total population of African and Asian cities is predicted to grow more than tenfold – from 309 million to 3.9 billion (Ramalho and Hobbs 2012).

Urban areas are heterogeneous in relation to biodiversity and NCPs, through a variety of natural, altered and novel habitats that support varied animal and invertebrate species. Fertile soils in urban areas enable urban residents to grow food ($\approx$ 15-20% of the world's food: Armar-Klemesu 2000), and green spaces provide recreational, cultural and health NCPs (Gomez-Baggethun et al. 2013).

Urban areas are usually rich in non-native species, whether naturalised or maintained in gardens, and extension occurs usually into agricultural more than natural land. Vegetation in urban areas often has enhanced growth relative to matched rural settings (Zhao et al. 2016). Land conversion is greatly reducing the extent of green space within many of the world's cities (Bagan and Yamagata 2014). At low levels of urban development, local species numbers may increase due to habitat heterogeneity (McKinney 2002). Non-native species may predominate in larger than smaller settlements (as many as 50% of species in a city centre can be non-native) and accumulate over time (Muller et al. 2013). Biotic homogenization increases along rural-urban gradients with city centres featuring "global homogenizers" - weeds, pests and commensals. Disease organisms and parasites can become abundant in urban systems, through the large reservoirs of animal (e.g., rats, bats, birds, foxes – Hassell et al. 2017) and human hosts.

Attribution of trends to drivers of varied species denisties can be difficult because of legacies of previous land use, transient dynamics, and few studies consider all the relevant drivers (Ramalho and Hobbs 2012). The main direct driver is replacement of vegetation by impervious surfaces. In the US, domestic cats (mostly feral) kill 1.3-4.0 billion birds and 6.3-22.3 billion mammals per

year (Loss et al. 2013). Pollution in urban areas is omnipresent, with nutrients and trace metal elements coming from residential, commercial business and industrial complexes (Sustainability & 2010). Waste treatment within or close to urban areas is a big driver of ecosystem change and threat to freshwater and animal species.

Phenotypic evolution is accelerated in urban landscapes compared to natural or agricultural ones (Alberti et al. 2017), as species adapt to novel and rapidly changing conditions. Urban ecosystems can provide insights into some aspects of climate change – cities tend to have higher temperatures because of the heat island effect, higher CO^2 levels and higher nitrogen deposition (Zhao et al. 2016).

Cities and municipalities have embarked on restoration of ecosystems, such as species diversity enhancement, or conversion of sewerage treatment plants to natural systems of waste treatment, filtering and purification (Allison & Murphy, 2017). In some city-regions, tree-planting as a restoration drive is combined with social interventions to create economic opportunities and address poverty (Mugwedi et al. 2017).

2.2.7.11 Cultivated areas

Cultivated systems are areas in which at least 30% of the landscape is in farmland or confined livestock production and managed for food/feed production. Globally 80% of the 1.6 billion ha of cultivated lands are rainfed; 20% occur in marginally suitable areas (FAO, 2011a). Further, 43% of cultivated lands are considered as agroforestry systems with more than 10% tree cover (Zomer, 2016). These cultivated systems are vital for sustaining food production and meeting the food and nutritional needs of growing human populations projected to exceed 9 billion people by 2050 (FAO,2017). The world's cultivated area has grown by 12% over the last 50 years, trebling the agricultural production (FAO, 2011a) to meet food demands.

Cultivated systems can themselves be degraded through human actions, and agriculture has the potential to have massive irreversible environmental impacts (Tilman et al. 2001). The combined impact of land degradation, desertification and drought affect more than 1.5 billion people in 110 countries, 90% of whom live in low-income areas (FAO, 2011a). Excessive use of fertilisers and pesticides have exacerbated land and soil degradation and erosion, although appropriate soil conservation practices that reduce erosion, such as minimum tillage, are increasingly being adopted by farmers (Derpsch et al 2010). There exist also many good examples of positive interactions between agriculture and biodiversity in agroforestry systems, species-rich meadows and other managed cultivated systems with biodiversity objectives in mind.

Land conversion of natural ecosystems to agriculture continues to be a major issue. Between 2000 and 2012 global oil palm planting area has expanded from 10 to 17 million ha (Pirker et al., 2016). A new paradigm, sustainable intensification (SI), is now emerging to grow more food more intensively, based on the need for increasing productivity while increasing environmental sustainability (FAO,2011b; Garnett et al 2016; Biodiversity International 2017).

Globally livestock production is the largest user of agricultural land and therefore also leaves a significant imprint on the environment (FAO, 2015a). Data suggest that there are large differences between production systems and type of livestock and demonstrate the importance of grasslands as a global resource (Herrero et al., 2013).

Key drivers negatively affecting cultivated areas include climate change: IPCC (2014) predicts that climate change will reduce agricultural production by 2% every decade while demand will increase by 14% every decade until 2050. Up to 40% of the world's land surface will develop novel climates, often with new pest and weed complexes (Lobell and Field. 2007). Pollution: there is evidence that the use of toxic agrochemicals and systemic pesticides, such as neonicotinoids, in cultivated systems is affecting non-agricultural lands and wild biodiversity including pollinators and other beneficial organisms (Dudley et al, 2017). Invasive alien species: transboundary pests and diseases are resulting in total crop failure and affecting the productivity of cultivated systems. Globally, annual crop losses to plant pests are estimated to be between 20 to 40 percent of production (FAO, 2017). These drivers will negatively impact the capacity of cultivated systems to continue to provide food and feed and to ensure the sustainability of food and nutritional security of human populations in decades to come.

2.2.7.12 Cryosphere

The Cryosphere is comprised of all locations on Earth with frozen water, including the Arctic, Antarctic, and glaciated mountain ranges within the polar regions. It stores about 70% of the world's freshwater as ice (Gleick, 1996), helps to radiate energy back out to space with its high-albedo surfaces, and is home to many extremophiles (Thomas and Dieckmann, 2002). This region contains fewer, larger, and more-complex organisms than temperate and tropical ecosystems.

The Cryosphere contains many unique ecosystems: Ice sheets, glaciers, and ice shelves contain all of the terrestrial, and terrestrially connected, ice on Earth. This land ice provides fresh water into adjacent ecosystems during melting events. The ice is home to extreme microbes living within thin water veins between ice grains. Sea ice covers portions of the Arctic and Southern Oceans, varies in extent seasonally, and provides shelter and hunting opportunities for many polar animals including polar bears, seals, penguins, and orcas. Extreme deserts, such as the Antarctic Dry Valleys, provide insight into the limits of life on Earth, and the types of microbial ecosystems that may be on other planets (Convey, 2006). Sub-glacial lakes found under ice sheets, such as Lake Vostok, Antarctica, are isolated systems where organisms have evolved independently for millions of years.

Climate change is having the greatest impacts on Arctic ecosystems, where warming has occurred at more than twice the global average during the past 50 years (Pithan and Mauritsen, 2014). Arctic land ice volume (Gardner et al., 2013), supporting ice shelves of the East and West Antarctic ice sheet (Pritchard et al., 2012; Hillenbrand et al., 2017), snow cover duration and

extent (Derksen and Brown, 2012), and sea ice thickness and extent are declining (Lindsay and Schweiger, 2015). The rapid warming is causing global sea level rise (Nerem et al., 2018), poleward and upward advancement of the treeline (Harsch et al., 2009), altering ranges of Arctic species including polar bears (Rode et al., 2012) and caribou (Vors and Boyce, 2009), altering animal diets (Rode et al., 2015), shifting predator-prey relationships due to phenological mismatches (Gilg et al., 2009), changing migration patterns of many species including anadromous fish (Mundy and Evenson, 2011), and desiccating terrestrial freshwater systems (Smol and Douglas, 2007). In the Southern Hemisphere, the strongest rates of warming are occurring in the West Antarctic Peninsula causing growth rates and microbial activity to rapidly increase (Royles et al., 2013). The Southern Ocean also continues to warm and freshen from increased precipitation and ice melt (Swart et al., 2018).

Sea level rise and severe storms have destabilised Arctic infrastructure, disrupting the physical, social, and cultural well-being of IPLCs (Cochran et al., 2013), and in some cases, forcing relocation (e.g. Alaska, Maldonado et al., 2013). ILK has been used in conjunction with Western science to further study the impact of climate change on Polar Regions (Pearce et al., 2015). Trends observed by IPLCs relate mostly to population trends such as reduced number of seals and increased population size of bears (Wong et al. 2016).

There are increased economic opportunities due to the increased number of ice-free days within the Northern Sea Route (Russia) and Northwest Passage (Canada), which will increase land- and freshwater-based transportation networks in the Arctic (Khon et al., 2010), bringing increased risk of ecological damage. The Arctic Council and its circumpolar Indigenous participant groups work to support research and legislation aimed at resolving issues surrounding sustainable development and environmental protection, through sharing of knowledge.

2.2.7.13 Aquaculture

Aquaculture converts terrestrial, freshwater or marine areas to farming of aquatic organisms, driven by depletion and stagnation of wild fisheries and rising demand and recognition of nutritional and sustainability benefits of aquaculture (Pelletier et al. 2011, Troell et al 2014a, Waite at al. 2014, Munkung et al. 2014). Estimates of global area of biomes converted to aquaculture does not exist – only sporadic national statistics (Ottinger et al. 2016). Freshwater fish from ponds makes up 60% of global aquaculture production, marine mussels and oysters 21 %, shrimps and other crustacean from ponds 10 % and marine finfish (mainly cages) 8.5 % (FAO 2018). Farmed seaweed production reached 30 million tonnes in 2016 (FAO 2018). China, India and Southeast Asian countries represent 80% of global aquaculture production (FAO 2018), followed by Bangladesh, Egypt and Norway.

Aquaculture production is projected to grow 15-37 percent by 2030 (Worldbank 2013, Kobayashi et al. 2015, FAO 2018), led by currently dominant species and countries (Hall et al. 2011, FAO 2018). Expansion faces challenges related to environmental impacts and competition for resources, e.g. feed, freshwater and energy (Bostock et al., 2010, Troell et al. 2014b, Pahlow

etal. 2015, Gephart et al. 2017, FAO 2018). Access to space will be an issue for land and coastal farming but not for off-shore ocean aquaculture (Klinger et al 2017, Troell et al. 2017a, Oyinola et al. 2018).

Aquaculture is the fastest growing food sector contributing 80 million tonnes (53 percent) to global food fish production (FAO 2018). Although 600 freshwater and marine species, across multiple trophic levels and culture techniques, are farmed worldwide, about 20 species comprise 84 percent of total aquatic animal production (FAO 2018). The value of mariculture products reached 65 billion USD in 2013, or 43 percent of global aquaculture (Oyinola et al. 2018).

Sustainability of culture species and systems varies widely (Klinger and Naylor 2012, Troell et al 2014a, Henriksson et al. 2015, Gephart et al. 2017). Today 70% of total animal aquaculture production relies on supplemental feed (FAO 2018) derived from a wide variety of food-quality and human-inedible sources, with important repercussions on the resilience of the world's food systems (Naylor et al. 2009, Tacon et al. 2011, Troell et al. 2014a, Troell et al. 2014b, Tacon and Metian 2015, Froehlich et al. 2017).

Climate change and global change, including unfavorable temperature regimes, hypoxia, sea level rise, ocean acidification, floods, diseases, parasites and harmful algal blooms and freshwater shortage (Myers et al., 2017, Barange et al. 2018) challenge aquaculture production. Antimicrobial use in aquaculture is also a cause of concern in relation to antimicrobial resistance (AMR) (Rico et al. 2012, Henriksson et al. 2018, Han et al. 2017).

Aquaculture can contribute to the global sustainability goals by providing incomes and supporting food security – especially in low- and medium-income countries (Little and Bunting 2016, Bené et al. 2016, FAO 2017). Farmed fish and shellfish are high in protein and rich in micronutrients, and employment is created throughout the aquaculture value chains (Bostock et al., 2010, Beveridge et al. 2013, Bené et al. 2016). However, corporate and community aquaculture have very different benefit sharing outcomes, particular for the poor. This requires appropriate policy development in producer countries.

2.2.7.14 Inland waters

Inland waters are permanent water bodies, including all types of lakes independent of salinity and depth, rivers, streams, ponds, water courses, cave waters). Declines in biodiversity of fresh waters are greater than those in the most affected terrestrial ecosystems (Dudgeon 2005; Sala et al. 2000). In Europe, 59% of freshwater molluscs, 40% of freshwater fishes and 23% of amphibians are threatened with extinction, due to chemical stressors, climate change and UV-B radiation (IUCN 2017). Freshwater species populations suffered an 81% decline (WWF 2016).

Total diversity of fresh waters is far from being completely studied (Gatti 2016). 115-188 new amphibian species were described annually between 2004-2016 (AmphibiaWeb 2017). Since 1976 around 305 fish species have been described annually (Reid et al. 2013). Lake Ohrid is a

major European biodiversity hotspot, characterized by its narrow endemism, however this is under threat from a wide range of anthropogenic pressures (Kostoski et al. 2010).

Flow modification is a particular risk for river ecosystems degradation. Dams change turbulent flowing waters to still, creating unfavourable conditions for specialist and endemic species and altering assemblages of taxonomic groups (Liermann et al. 2012). Retention of water in dams is as high as five times the volume of all the world´s rivers (Nilsson and Berggren 2000). 172 out of the 292 large river systems are affected by dams, with Europe having the smallest number of completely unfragmented river systems (EEA 2015; Nilsson et al. 2005; Sanz and Rubail 2016). The Mekong, Congo and Amazon are the most biodiverse river basins on Earth affected by dam construction (Winemiller et al. 2016).

Global environmental changes such as nitrogen deposition, climate change, shifts in precipitation and runoff patterns (Galloway et al. 2004) affect inland waters, and are superimposed upon other localized threats (Dudgeon et al. 2005).

Biodiversity losses can affect water quality, e.g, by loss of species that remove excessive nutrients (Cardinale 2011). Populations of different important fish species declined significantly, while introduced species transform the original fish communities (Aigo and Ladio 2016, Gray et al. 2017).

2.2.7.15 Shelf systems

Shelf systems extend from the shoreline to 200m deep, comprising 8% of the earth's surface (Kaiser et al. 2011) and contribute 90% to the world's marine primary production (Longhurst et al. 1995). They are influenced by adjacent terrestrial systems and watersheds; urban, aquaculture and intensively used coastal areas; and in polar regions, the cryosphere. This makes shelf ecosystems among the most vulnerable to cumulative and intensifying local to global impacts.

Shelf systems comprise several sub-units: mangrove forests and seagrass beds are dominated by flowering plants adapted to salt water. Both sequester more carbon than tropical rainforests. Coral reefs flourish in shallow tropical seas due to symbiosis between hard corals and intra-cellular dinoflagellates. Other biogenic reef habitats are created by e.g. tubeworms, bivalves, and sponges. The intertidal zone, comprising rocky and sandy shores, is controlled by physical extremes and aerial exposure in upper levels, while ecological interactions dominate at lower levels. Macroalgal habitats become more dominant at higher latitudes, with giant kelp reaching heights of 40 m. Submerged habitats on the shelf include rocky, cobble, sand and muddy bottoms, which determine the biological communities they support. Deep coastal inlets and fjords support concentrated diversity hotspots. Polar shelves with poorly sorted sediments especially in the Southern Ocean support unusually high biomass of heterotrophs (Gutt et al. 2013). Coastal pelagic areas include highly productive waters where plankton are the primary and secondary producers and sustain rich fisheries yield, such as polar seas, the North Sea, Sea of Okhotsk and East China Sea.

Shallow shelf ecosystems have supported human uses for tens of thousands of years as a result of their accessibility and high productivity, for fishing, natural products, tourism and coastal development. Cumulative impacts are evident (Selig et al. 2014). Global cover of mangroves (134,000 km^2) has declined 37.8% (Thomas et al. 2017). Shallow coral reefs have shown long-term decline (Pandolfi et al. 2003) and are losing live coral cover at a rate of 4% per decade (Section 2.2.5.2.1); severe global bleaching events are increasing in frequency and intensity because of rising temperatures (Hughes et al. 2018). Conditions currently unsuitable for persistence of shallow coral reefs globally are predicted to occur within the next 10-50 years at almost all reef locations globally (van Hooidonck et al. 2016, Beyer et al. 2018), and >33% of coral species are listed as Threatened (Carpenter et al. 2008). The reef- associated fish species Living Planet Index (LPI) declined 34 per cent between 1979 and 2010 (WWF-ZSL, 2015).

Drivers of shelf ecosystem decline include fishing, eutrophication, solid and liquid waste, habitat fragmentation, underwater noise from shipping and invasive species. Indirect effects of land-use change are mediated through freshwater runoff from land and in rivers. Climate change is increasingly pervasive in shelf systems (Hoegh-Guldberg et al. 2014), through increasing temperature, acidification, deoxygenation and intensifying storms. They fundamentally affect species' life histories, as well as the physical structure of the coastline and shelf.

Shelf ecosystems are of great significance to IPLCs. Many coastal cultures have detailed histories and mythologies related to them (Lee 2014), as well as centuries and even millennia-old practices and customs demonstrating intimate adaptation (Johannes 1981). However, the commercial over-exploitation and decline of many shelf ecosystems contributes to the loss of these traditions. Both IPLCs and scientists document the decline in abundance of fish species (e.g., sawfish species in Brazil) and weight of fish (e.g., goliath grouper) (Giglio et al. 2015, Reis-Filho et al. 2016).

Shelf ecosystems are an increasing focus for management and protection. Marine Protected Areas and sectoral tools (e.g. in fisheries, shipping, etc) are now being integrated into novel approaches including Integrated Coastal Zone Management (Clark 1992) and Marine Spatial Planning (Ehler and Douvere 2009). Direct and spatially explicit conservation and protection measures are generally local, though increasingly applied at scale as countries approach 10% targets for marine area management. Improving the effectiveness of management is recognized to be equally important as area, to assure benefits accrue to users (Cinner et al. 2016, Edgar et al. 2014).

2.2.7.16 Surface Open Ocean

The Surface Open Ocean is the shallower light-flooded layer offshore of the 200-m depth contour (Fig. X). It covers 65% of the earth's surface (Kaiser et al., 2011), converts regionally high amounts of carbon and nutrients to biomass, and remineralises more than 95% of this organic matter (Ducklow et al., 2001). The Surface Open Ocean and shelf ecosystems produce 50% of atmospheric oxygen (Field et al., 1998) and sequester anthropogenic CO_2, which is

essentially important for almost all life on Earth. This function is expected to weaken with increasing climate change. Biological processes in the Surface Open Ocean are driven by sunlight, nutrient availability, and water mass stratification. The unit exchanges with the deep-sea through downward flux of organic matter, upwelling of nutrients and vertical migration of organisms.

The Surface Open Ocean comprises different ecosystems: <u>Central Oceanic Gyres</u> contribute to the global dispersal of heat, nutrients and organisms. They include oligotrophic 'deserts' and highly productive areas (Westberry et al., 2008). <u>High-Nutrient Low-Chlorophyll Systems</u> occur in the Southern Ocean, the subarctic and equatorial Pacific Ocean, where phytoplankton growth is not limited by macronutrients (Pitchford and Brindley, 1999). <u>Cold and Ice-Covered Polar Seas</u> are driven by high seasonality and low temperatures. Their productivity supports krill (Atkinson et al., 2008), which feeds penguins, seals, and whales that migrate across the oceans. <u>Upwelling Systems</u> allow high fishing yields based on high primary production (Kämpf and Chapman 2016). <u>Oxygen Minimum Zones</u> are caused by excess carbon decomposed by bacteria with anoxic metabolism (Karstensen et al., 2008; Levin, 2003).

Due to its size, the Surface Open Ocean is still poorly characterized in spite of centuries of ocean voyages and expeditions. Its approximately 7000 species are less than in some coastal systems and the deep-sea (Bucklin et al., 2010). Hotspots in species richness are for example in the marginal seas of Southeast Asia and polar regions.

The Surface Open Ocean is vulnerable to threats, including from fisheries, pollution including waste, shipping, and noise. Environmental changes have been documented in ocean circulation and chemistry, thermal stratification, composition and growth of phytoplankton (Sarmiento et al. 2004; Boyce and Worm 2015), biogeochemical cycling (Hoegh-Goldberg and Bruno 2010; O'Brien et al. 2017), and distribution of ecologically key species (e.g. Beaugrand 2009) with effects on food webs (Smith et al., 2008; Knapp et al., 2016). Fishing has altered trophic relationships (Pauly 1998; Richardson et al. 2009), the number of overexploited fish stocks, e.g. of tuna and billfish has increased over the past decades resulting in regionally declined fishing yields by 50% (Worm et al. 2005; Sherman and Hempel 2009). Waste accumulation is documented though poorly known (Bergmann et al. 2015). Extinction risk for open ocean species has been assessed for seabirds, tuna and sharks (Brooks et al. 2016).

The ocean surface is sensitive to climate change, experiencing a globally averaged 0.44°C warming between 1971 and 2010 (IPCC 2014). Ocean acidification affects not only key calcifying pelagic organisms, such as pteropods and coccolithophorids, it potentially changes the physiology of all species (e.g. Manno et al. 2007).

Interactions of IPLCs with the Surface Open Ocean includes the historic navigation of Micronesian and Polynesian seafarers (Lee 2014) and is found in notes of captains of fishing vessels, whalers, and explorers (Holm et al. 2010; Rodrigues et al. 2016).

Protective management of Surface Open Ocean systems is increasing as they become less remote with modern technology, trade and extension of governance regimes. In spite of increased pressure, the number of sustainably managed fish stocks has increased (FAO 2014; MSC 2016). Targeted species such as Antarctic fur seals and Humpback whales are recovering (Zerbini et al. 2010) and strategies to reduce by-catch by longlines and driftnets of e.g. turtles, albatrosses, and dolphins are being developed (Hall et al. 2000; Kennelly 2007). The area of ocean under protection is expanding with accelerating designation of Marine Protected Areas and development of legally binding instruments for governing the High Seas (Wright et al. 2016).
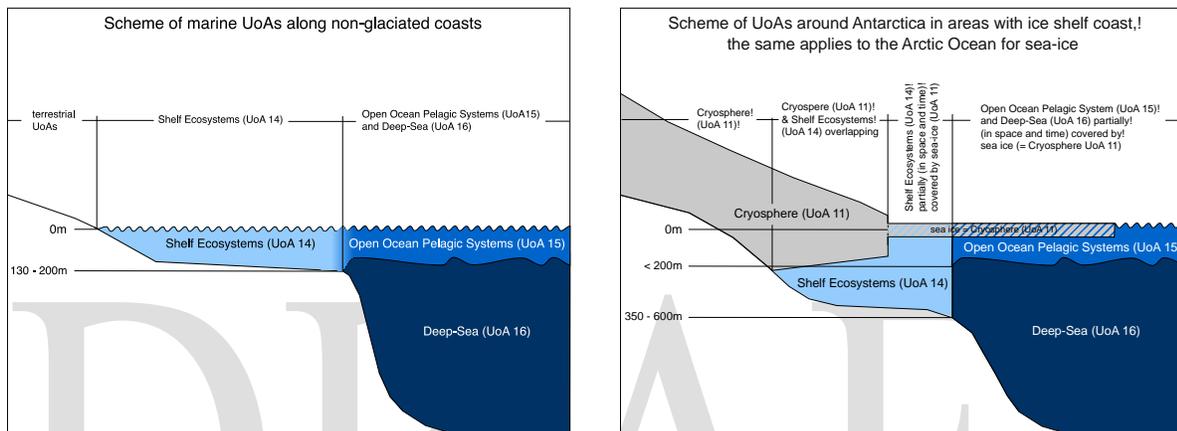
Figure 2.26 Left- the ocean units (14, 15 and 16) are distinguished across biological and environmental gradients rather than by discrete differences in water masses, nutrient supply, sediment characteristics, and species assemblages. The boundary between units 15 and 16 is usually denoted by the compensation depth, above which primary production happens, and set by convention at approximately 200 m depth, but this can vary over space. Right - the relationship between shelf (unit 14) and open ocean (unit 15) units and the crysophere (unit 11) is complex, as ice layers over water and may be land-fast or free-floating. Further, the boundary between units 14 and 15/16 is at the 200-m depth contour around most continental shelves, down to 500 m in Antarctica.

2.2.7.17 Deep sea

The deep-sea is the largest and most three-dimensional habitat on Earth. It comprises the dark waters below the euphotic zone (200 m, Fig. X), where biological processes remineralise nutrients and sequester carbon, including of anthropogenic origin, as well as other ecologically important elements. Almost all life in the deep-sea depends on climate-sensitive biological processes in the surface layer (Smith et al. 2008) and in the sea-ice (unit 11). Through the globally connected current system damage to deep-sea ecosystems, especially by pollution, affects natural resources directly used by man.

The deep-sea comprises a number of components: the Slope and Rise of Continents and Islands from 200 to 4000 m depth, are characterised by steep and diverse environmental gradients and peak benthic species richness between 1500 and 3000 m (Ramírez-Llodra et al. 2010). The Abyssal Plain from 4000 to 6000 m covers the largest area (more than 50% of the Earth's

surface), where due to limited food availability metabolic rates and biomass are low (Woolley et al. 2016). Faunistic depth gradients are superimposed by a decrease in species richness from the equator to the poles (Ormond et al. 1997). One of the most speciose bottom dwelling animal groups are nematodes (e.g. Danovaro et al. 2010), whilst bacteria perform highest biological turnover rates. The Mid-Ocean Ridges are created by seafloor spreading, with peaks between 5000 and 2500 m above the abyssal plains. Their complex topography and variable sediments shape sea-bed and pelagic assemblages (Vecchione et al. 2010). Vents and Seeps provide energy in the form of methane and sulphides, driving chemosynthetic food webs based on specialized microorganisms (Baker et al. 2010); similar communities develop on whale falls. Some vents provide clues to the deep biosphere living within deep sediments and the ocean crust (Schippers 2016). Seamounts rise more than 1000 m above the surrounding seabed (Clark et al. 2010), where upwelling of nutrients increases biological productivity. Their sessile benthic filter feeding biota is highly endemic (Richer de Forges et al. 2000). Seamount productivity supports abundant fishes, sharks, turtles, marine mammals and seabirds. Deep-Water Coral Reefs create a three-dimensional habitat for a rich associated fauna without light-enhanced growth (Freiwald et al. 2004). Deep-Sea Trenches occur between 6000 and 11,000 m depth, with a fauna low in abundance and biomass. Life in the Deep Aphotic Pelagic Zones, including the meso-pelagic twilight zone (200-1000 m, Sutton et al. 2017), and the bathyal, abyssal, and hadal zones (1000-11,000 m) mostly depends on organic matter falling from the light-flooded surface water layers. It comprises gelatinous invertebrates and midwater fish adapted to a stable environment (Ramírez-Llodra et al. 2010).

The low abundance of organisms and low scientific sampling in the deep-sea and an assumed high proportion of range-restricted species make species numbers hard to assess, but it is thought rival other global biodiversity hotspots (Knowlton et al. 2010).

Anthropogenic damage in the deep-sea is less than in shallow waters and on land but is increasing rapidly. A severe impact is bottom trawling (Clark et al., 2016) on fish (e.g. grenadiers and orange roughy), resulting in rapid decline of yields of slow growing species after a short phase of overfishing and damage to unique benthic habitats, especially on seamounts. Deep-sea mining is expected to be a major threat in the near future (Jones et al. 2017). Long-term effects of dumped waste, especially radioactive material and plastics is still largely unknown (Bergmann et al. 2015). Due to adaptation to a stable environment most deep-sea organisms are sensitive to environmental changes, especially to climate-induced shifts in energy supply, alteration of biogeochemical cycles including ocean acidification and prey-predator interactions.

Historic Indigenous knowledge on deep-sea organisms is common to many ancient seafaring cultures, in the form of tales of mythical bizarre creatures from an unknown world (Ellis 2006). Conservation of deep-sea habitats is still rudimentary and sectoral, but concepts for ecosystem management and Marine Protected Areas to reduce the impact of bottom trawling (Wright et al. 2015) and deep-sea mining (Wedding et al. 2013) exist.

2.2.7.18 Coastal areas intensively and multiply used by humans

The coastal area includes the coastal waters, the seabed, adjacent land and nested waterbodies (including freshwater). Coastal areas extend along more than 1.6 million km of coastline in a total of 123 countries (UNEP 2006b). At present a third of the world's population is living in the coastal zone and almost 40% of the world lives within 100 km of the coast (Agardy et al. 2005).

Coastal areas are experiencing an intensification of multiple uses, due to human population growth, migration from inland regions, tourism and economic growth. Coastal land is used for human settlement, agriculture, trade, industry and amenities. The coastal sea is intensively used for transport, fishing, dumping, mining, and more. Furthermore, coastal areas are the "sink" for the continents; they receive and concentrate pollutants and other negative consequences of anthropogenic activities. Carbon cycling in the coastal sea that connects terrestrial with open ocean systems plays an important role in the global carbon cycles and budgets (Regnier et al. 2013). Tourism is a very important driver in many regions and is responsible for a great increase of pressure in coastal areas. Continued human uses and pressures in coastal zones will have an important impact on the future evolution of the coastal ocean's carbon budget.

Coastal areas intensively and multiply used by humans is an anthrome, defined by artificial constructions linked to human settlements, industry, aquaculture, or infrastructure that transforms coastal habitats (Bauer et al. 2015). These include a) coastal defences (breakwaters, groynes, and jetties), b) coastal protection (seawalls, bulkheads, and pilings), c) floating docks, e) artificial islands, f) dumping and mining areas, g) artificial structures for energy (including renewable energies) and h) port development and coastal support. Population growth, industrial and tourist development, pollution, habitat and biodiversity loss, changes in access rights, markets and technology and increasing drivers of global change are threatening the future sustainability of coastal areas. Although many of these changes occur in other ecosystems, they are particularly concentrated on the coast.

People living in the coastal areas and particularly poor coastal communities, have adapted to transformations in coastal ecosystems. But now they face an environment of increased competition from high-density and industrial uses, in which access to the resources they depend on is becoming more and more restricted. Additionally, future sea-level rise is also putting pressure on coastal areas. Coastal management needs to encompass decisions of which uses to regulate, which uses to promote, and which NCPs are most important to citizens and businesses (Loomis et al. 2014) to provide for sustainable use of the resources of the coastal areas, by addressing trade-offs between conflicting multiple uses.

There is an urgent need for a holistic coastal zone management approach (integrated, multiple-use oriented) to provide mediation through administrative procedures, public hearings and facilitated dialogue, for stakeholders (including coastal communities and local and central governments) to be represented in negotiations. Strengthening the integration of IPLCs and ILK in multiple use planning and management in the coastal areas is essential to long-term sustainability of coastal areas (Lockie et al. 2003).

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