5. Modelling consequences of change in biodiversity and ecosystems for nature's benefits to people

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Key Findings

The main contribution of an ecosystem service approach to decision making comes from considering bundles and tradeoffs among multiple ecosystem services. Assessments of an ecosystem service in isolation can be useful for specific contexts, but assessing ecosystem services individually risks hiding tradeoffs and synergies between ecosystem services that are often crucial in many decision-making contexts.

Modelling methods, tools, and participatory processes each have particular strengths and weaknesses that make them better fits to different decision contexts. This chapter provides guidance on how to match tools and decision contexts. Complex models are useful for integration and large-scale analysis, but in many contexts relatively simple models can be more useful than complex models, because they are easier to understand, use, and assess. Simplicity is especially important in assessing multiple ecosystem services where the reliable models of multiple services have not been well developed.

Applying multiple models to the same case produce more robust decisions because applying models with different strengths and weaknesses can provide a more complete picture and comparing model results can indicate where models provide variable or consistent results. This chapter explains how different types of models can effectively complement one another.

Models of ecosystem service are undergoing rapid development. The number, diversity and application of ecosystem service models has greatly increased over the past decade. A variety of ecosystem service models exist, however most models are have limited ability to explain dynamic processes or social-ecological feedbacks, consequently the ability of ecosystem service models to project or analyse alternatives is low. Many new types of ecosystem service models are in

development, and research now aims to address many of the limitations of ecosystem service models over the next decade.

Modelling the impact of ecological changes on human well-being is not well developed. Developing such tools will require investment and trans-disciplinary collaboration of policy makers, with natural and social scientists to develop new frameworks, methods, and tools. The development of models that integrate different ways of assessing human well-being is particularly needed as there are a diversity of ways in which human well-being can be assessed, and the study of human well-being is also rapidly developing.

Models of biodiversity and models of ecosystem services are not well connected. Ecologists increasingly understand how biodiversity produces ecological functions (Chapter 4), however most models of ecosystem services utilise land use and land cover to predict ecosystem services. Including biodiversity in ecosystem service models is challenging due to a lack of spatially explicit biodiversity data. Land use and land cover are related to biodiversity, but spatial configuration, history, and management also shape local and regional biodiversity. Making progress on the connections between biodiversity and ecosystem service models would improve ecosystem models, as would improving social and abiotic factors. Because ecosystem services are produced by social and ecological factors in addition to biodiversity, so including all these aspects of would likely increase the predictive quality of ecosystem service models. Which approaches yield the biggest improvements in model quality will likely depend upon social-ecological context, data availability, and the ecosystem service being considered.

Key recommendations

We recommend that IPBES should foster the development of a community of practice for ecosystem service modelling. Such a community of practice requires programmes of training, flexible standardisation, open access to data and models. Building this capacity is important for enhancing tool development based on diverse knowledge systems, tools specialised for different decision and geographical contexts. There is substantial experience within the scientific community in developing such communities, and this expertise and experience should be utilised. IPBES should use its Knowledge, Information and Data task force (Deliverable 1d) to facilitate access to models and data by encouraging governments and scientists to make their models and data freely available using open access or creative commons licensing. IPBES should use its Capacity Building task force (Deliverables 1a and b) to promote a community of practice for ecosystem service modelling to advance ecosystem service modelling as well as encourage the development of new frameworks to assess the relationship between people and nature.

We recommend that IPBES should foster the development of a community of practice around scenario building practices. Participatory social-ecological scenario have been increasingly used to explore ecosystem services in alternative futures. While such scenarios enable diverse and

qualitative knowledge about ecosystem services to be combined with quantitative models, it is currently difficult to compare and build upon specific scenario processes as they are wedded to particular people, times, and places. A community of practice that uses common methods, and addresses shared issues would increase the ability of scenarios to bridge across scales and cases. IPBES should use its Knowledge, Information and Data task force (Deliverable 1d) and Capacity Building task force (Deliverables 1a and b) to promote such a community of practice to enhance the ability of people to use, develop,

We recommend that IPBES promote developing new ways to include multiple values and indigenous and local knowledge systems in models and scenarios. Alternative values, multiple knowledge systems, and indigenous and local knowledge are rarely addressed in current modelling work, yet have been highlighted as a priority area for IPBES. If these issues are to be explored in regional and global assessments there will have to be investment in including multiple values and knowledge systems in models and scenarios. IPBES should ensure that the task forces on Capacity Building (1a & b), Indigenous and Local Knowledge (1c) and Knowledge, Information and Data (1d) and the expert group on Values (3d) C facilitate communication among these communities as well as the development of new model and scenario approaches.

We recommend that thematic, global and regional assessments of ecosystem services (IPBES Deliverables 2b & c, 3b) analyse outputs from models of ecosystem services at multiple scales. In particular, global and regional models that evaluate multiple ecosystem services are recent developments. They have not yet been sufficiently tested and often do not correspond to ecosystem services observed in many places. Local scale models of multiple ecosystem services have been much more widely tested and applied, but methods for scaling up to regions and or the globe pose many challenges. We recommend that global and regional assessments link and analyse connections among multiple cross-scale ecosystem service assessments that include models of local ecosystem service dynamics.

We recommend that regional assessments of ecosystem services (IPBES 2b) link and analyse connections among multiple cross-scale ecosystem service assessments that use better developed models of local ecosystem service dynamics. Local models of ecosystem services are better developed than regional or large scale models of ecosystem services. Therefore we recommend that regional assessment should integrate and compare multiple local models of ecosystem services rather than rely on regional models of ecosystem services.

5.1 Introduction

Research on modelling the benefits that nature supplies to people, or ecosystem services, has rapidly expanded and diversified over the past decade. This chapter assesses the current state of these models from the perspective of IPBES. The first part of this chapter provides critical reviews of the key conceptual components in modelling connections between ecosystem services and human well-being , as well as how these connections are shaped by changes in biodiversity, anthropogenic assets, institutions, and other drivers (Figure 5.1). The second part of the chapter then reviews the main modelling approaches for assessing ecosystem services, and relates these approaches to the different decision contexts in which these models can be used. This chapter concludes with an assessment of gaps, recommendations for actions, and future research that would develop the capacity to better use of ecosystem services and human well-being models in IPBES.

5.2 Conceptual framework and knowledge for modelling ecosystem services and human well-being linkages

Ecosystem services and their contribution to human well-being are shaped by the combination of ecological structures and processes as well as social institutions and anthropogenic assets. Human well-being and nature are also directly impacted by social activities or conditions as well as biophysical dynamics (Fremier et al 2013, Butler & Oluoch-Kosura 2006). The IPBES conceptual framework (Figure 5.1) integrates these interactions. This section reviews current concepts that link ecosystem services to human well-being. The first part of this section briefly presents the current insights on the importance and applications of modelling ecosystem services and human well-being linkages. The subsequent parts of this section provide links between this chapter and the rest of the other chapters of the IPBES methodological assessment deliverable 3c. The remaining parts of this section, review the advances in research in relation to various important conceptual foundations for modelling ecosystem services and human well-being linkages and dynamics.

5.2.1 Ecosystem services, human well-being and the IPBES framework

Nature, including both biodiversity and ecosystems, provides goods and benefits to human societies. These are throughout this chapter referred to as ecosystem services (e.g., provisioning, regulating, cultural, and supporting services), that contribute to wealth (anthropogenic assets) and well-being of human societies (MA 2005). However, with the intensification of biodiversity exploitation, ecosystem conversion, species invasion, and climate change, nature's capacity to continuously provide sufficient benefits, wealth, and well-being to human societies is greatly reduced and compromised (Cardinale et al 2012). However despite the simplification, conversion, and degradation of many ecosystems, the past century has seen consistent and global increases in human well-being , as measured in terms of the human development index and many other measures. Much of the global simplification of ecosystem has been to replace diverse ecosystems with those producing high levels of agricultural ecosystem services, and to date this conversion has

enhanced rather than decreased human well-being (Raudsepp-Hearne et al 2011). In addition, even degraded ecosystems are known to provide some form of ecosystem services to beneficiary communities - such as diverse tropical forests converted to monoculture oil palm plantations (Abram et al 2014). Moreover, wild ecosystems many provide less of some services, such as disease regulation, than simplified ecosystems. Furthermore, during the past century social innovation and technical innovation have meant more efficient use of ecosystem services resulting increased human wellbeing, in terms of life expectancy, education, and income. However, while scientists have unravelled outlines of how ecosystem services contribute to human well-being, there how these multiple is a great need to assess the links between ecosystem services and human well-being using scenarios and models in order to be able to develop and implement policies that can help ensure the sustained flow of benefits from biodiversity to human society, and thereby contribute to human well-being . In addition, there is a great need to assess how social and ecological changes increase or decrease the supply, use and demand for ecosystem services various socio-ecological context. Moreover, modelling ecosystem services and human well-being linkages can provide valuable inputs in various deliverables of IPBES, especially with regards to various international targets such as the Convention on Biological Diversity's Aichi targets.

In the following sections we provides an overview of the key conceptual components in modelling connections between ecosystem services and human well-being , as well as how these connections are shaped by changes in biodiversity, anthropogenic assets, institutions, and other drivers (Figure 5.1). Other chapters in this assessment address other related parts of the IPBES conceptual framework. Chapter 2 focuses on the Decision contexts in which models of ecosystem services and biodiversity are used, Chapter 3 on Drivers of changes in nature and biodiversity, and Chapter 4 on Modelling impacts of drivers on biodiversity changes.

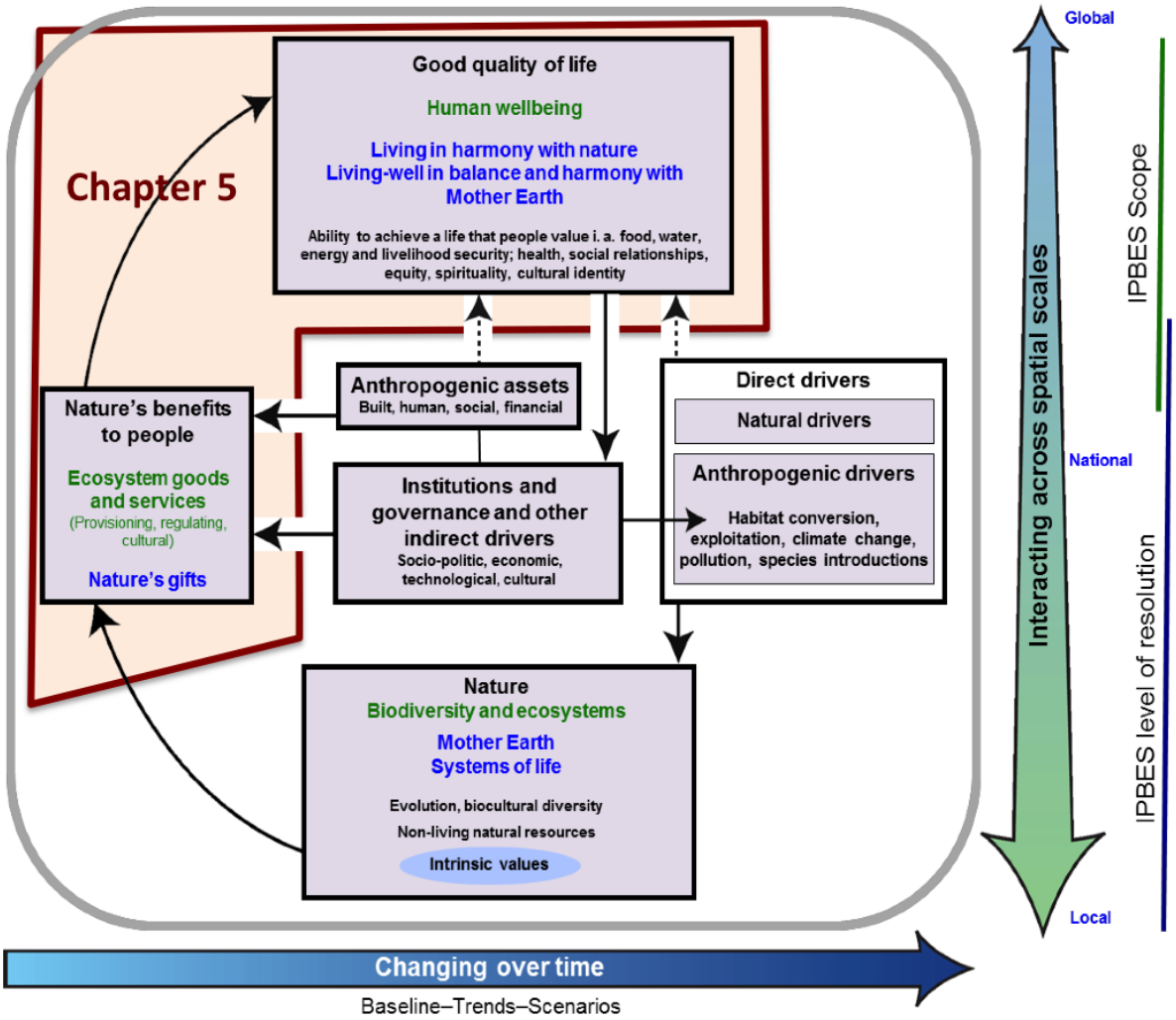


Figure 5.1.: The portion of the IPBES conceptual framework that this chapter focusses on modelling how nature provides benefits to people, and these benefits are influenced by nature, institutions, anthropogenic assets as well as natural and anthropogenic direct drivers.

5.2.2 Modelling ecosystem services and human well-being: needs, gaps, and proposed improvements

Modelling and scenario-building tools are important in untangling and understanding the complex interactions and feedbacks between ecosystem services and human well-being at various spatiotemporal scales. However, existing ecosystem services and human well-being modelling tools are limited in terms of the dominant economic valuation metaphor (e.g., cost-benefit analysis, willingness to pay, or benefit transfer), and need improvements by incorporating other nature-human relationships metaphors (e.g., humans as part of food-web, human's moral obligations with nature, or humans' spiritual relationships with nature) within a defined Social-Ecological System (SES) context (Raymond et al 2013, Reyers et al 2013).

In addition, ongoing changes and degradation of ecosystem services are happening on a large scale (e.g., climate change driven ecosystem and species range expansion, range collapse, or massive land

conversion and over-exploitation), of which actual data are often limited, and therefore benchmarking relationships between models and actual observed data will be difficult, and subsequently causing increase in uncertainty of most model and scenario projections (Carpenter et al 2009, Seppelt et al 2011).

Moreover, some studies suggest that existing modelling tools and frameworks for linking ecosystem services and human well-being need integrated social-ecological metrics or data that can actually capture flow of ecosystem services to various beneficiaries, and subsequently improve their well-being , such as final goods and services metrics, with the consideration of technology or conditions that enabled beneficiaries' access to ecosystem services (Ringold et al 2013, Daw et al 2011).

Therefore, improving collaborative (i.e., natural-social scientists collaborations) and empirical monitoring of important ecosystem services and human well-being indicators and metrics over a large scale (e.g., GEO-BON and LTER) should play a key role in improving current models of ecosystem services and human well-being relationships (Carpenter et al 2009).

5.2.3 Anthropogenic assets, ecosystem services, and human well-being

Anthropogenic assets in combination with nature can be understood as crucial sources of human well-being for present and future generations, since assets can in general be defined as tangible or intangible objects that yield, either directly or indirectly, inter-temporal flows of benefits to people (Dickie et al. 2014, Munoz et al. 2014 and UNU-IHDP and UNEP 2014). The key difference between assets and intermediate production flows, is that assets persist over time while flows are transitory. Understanding changes in assets is therefore a key part sustainability research.

The flows of benefits to people from nature, are commonly termed ecosystem services. The ecosystems that produce these services are often called natural capital. The ecosystem accounting literature has also referred to them as ecosystem assets (SEEA-EEA, 2014). While some ecosystem services can be accessed 'directly' by the final beneficiaries, as for example clean air, the enjoyment of some landscapes or recreational services; there are several other situations where people's benefits from nature can only be consumed when complementary anthropogenic assets are present (Burkhard et al. 2012). For instance, in the case of timber as a provisioning service, one can think not only of the machineries necessary to cut trees, but also the transport system (and roads) which support that the resource is available to final users or producers in the intermediate sectors for further processing. Another example of a complementary anthropogenic asset are vessels, being essential for fishing offshore and in remote areas. Such transport means, machineries, and infrastructure are commonly clustered within the so-called built or produced assets (System of National Accounts 2008). Moreover, produced capital is not the only anthropogenic asset utilised in the co-production of benefits by nature and society.

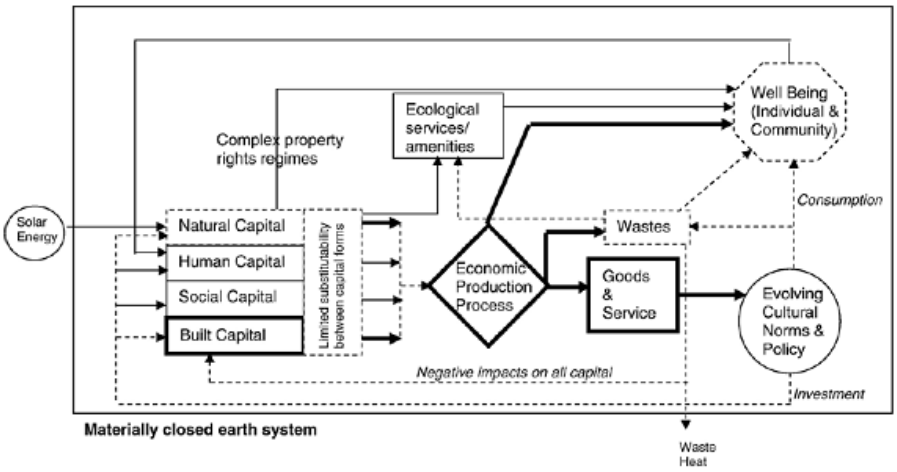
The different kind of knowledge, skills and abilities embodied in individuals (i.e., human capital) also contribute significantly to the production of ecosystem services. In this regard, one should think of knowledge in a broad sense, comprising indigenous and local knowledge systems as well as technical or scientific knowledge, including also formal and non-formal education (Díaz et. al. 2015).

Social capital also contributes to the production of ecosystem services. Social capital consists of social networks, norms, rules, interpersonal agreements, etc that enhance the ability of individuals to produce value (OECD 2001). While institutions can be considered as a form of social capital (OECD 2001), in the IPBES conceptual framework they are treated separately from anthropogenic assets.

These natural, produced, human, and social capital assets operate within an institutional framework that plays an important role for regulating and understanding the co-production dynamic by the different underlying factors (or anthropogenic assets) and the supply of ecosystem services. Ecosystem services modelling relies therefore on the combination of the *multiple* anthropogenic assets available in the society and nature (see figure 5.2) that make possible the access of the people to an ecological service.

Furthermore, making nature' benefits available to its final users is subject not only to the availability of complementary anthropogenic assets used in the provision of the service, but also to technology. In some instances, a resource or potential ecosystem services is known to exist, however, the current state of technology does not allow an efficient access to it as the costs of exploiting this resource exceed the willingness to pay for it. In such a case lack of appropriate technology can be seen as a constraint on the availability of ecosystem services. In other cases, however, technology may lead to an overexploitation of nature by the overuse of the resource, acting technology in this case as a driver of resource use (see next section and chapter 3). This is commonly articulated by substantial reductions in the prices of the goods and services resulting from nature that increase the demand for it (Giampietro and Mayumi 2008).

The comprehensive modelling of ecosystem services therefore requires the integration of ecological structures and dynamics, with social dynamic that include the interaction between anthropogenic assets, technological progress along with the institutional setting (norms, rules, political context , etc). All these factors govern the co-production of ecosystem services for human well-being. Indeed, the complex linkages that sometimes characterise the social-ecological dynamic requires that modellers extend the use of ecological production function to consider also the human dimensions, suggesting therefore the development of social-ecological production functions (Reyers et al. 2013). Moreover, it is recommended to capture in the modelling non-linear feedbacks, trade-offs, and drivers associated with services provision (Reyers et al. 2013). Important advances in the development of such integrated framework can be found in the Social-Ecological System (SES) approach (see e.g. Berkes et al. 2003, Ostrom 2009, and Reyers et al. 2013), where four core subsystems are usually characterised for analysing them: (i) resource systems; (ii) resource units; (iii) governance systems; and (iv) users (Ostrom 2009).



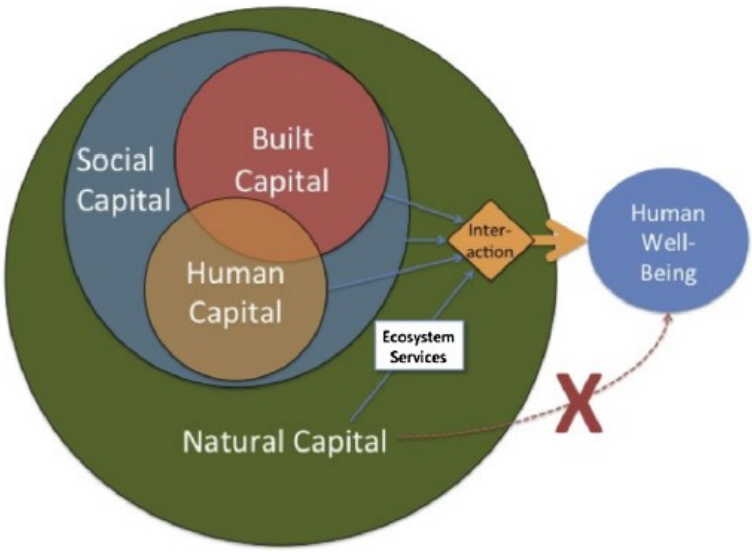


Fig. 1. Interaction between built, social, human and natural capital required to produce human well-being. Built and human capital (the economy) are embedded in society which is embedded in the rest of nature. Ecosystem services are the relative contribution of natural capital to human well-being, they do not flow directly. It is therefore essential to adopt a broad, transdisciplinary perspective in order to address ecosystem services.

Figure 5.2.:: There are many ways that natural and anthropogenic assets have been conceptualised as producing ecosystem services. Two representative examples are A) Natural capital being one of many factors of producing well-being (Walker and Pearson 2007), and B) Natural capital being the ultimate system in which all other capital are embedded (Costanza, et al. 2014). [Figures here will be revised to align with other chapters - suggestions for diverse frameworks welcome]

5.2.4 Institutions and other drivers of ecosystem services and human well-being

The flows of ecosystem services often accrue unevenly to various sectors of the society or beneficiaries, and are often influenced by direct anthropogenic drivers (habitat conversion, exploitation, climate change, pollution, population growth, species introduction, changes in income

and wealth, international trade, etc), direct natural drivers (e.g., earthquakes, volcanic eruptions, etc), as well as indirect drivers of change (institutions and governance systems, societal level of inequalities, cultural values and practices, policies, technology, etc). The exact relationships, interactions, and consequences of these drivers on ecosystem services and human well-being as well as the feedback loops and thresholds or tipping points need to be further explored and modelled. This section presents a brief overview of how institutions and other drives affect ecosystem services and human well-being . For a more detailed typology of drivers, see chapter 3.

Institutions: Institutions play an important role in the management and exploitation of Biodiversity and Ecosystem services (Abunge, et al. 2013; Christie & White, 2007; Lowry et al, 2005). Effective institutional design and implementation is however crucial. For instance, lack of clear delineation of responsibilities and conflicts affect coastal resource management. Inconsistencies and conflict in local vs. national plans and legislations for conversion and restoration of resources also has implications on ES, while weak enforcement of laws and regulations governing resources is also a challenge (Lowry, et al 2005). Common governance challenges include confused goals, conflict, and unrealistic attempts to scale up beyond institutional capacity. Property rights regimes influence management of resources. Where collective action and conflict resolution mechanisms break down, governance of ecosystem resources is compromised (Ostrom 1990). Socio-political, historical, and socio-economic context of a locality also influence governance of ecosystems (Christie & White 2007). Fragmented legal systems can lead to gaps and conflicts (Techera & Klein, 2011, Pomeroy et al. 2010), while governance of large scale ecosystems requires identification of the heterogeneous, multi-scale and interlinked nature of these systems (Fidelman et al. 2012).

Institutions can promote ecosystem services exploitation. For instance, in Thailand, policies that promoted shrimp farming by absentee landlords led to the massive destruction of mangrove ecosystems and thereby exposure of coastal communities to catastrophic storm and tsunami events (Barbier et al. 2011). Norms instilled by collective management under common property resources could regulate demand and utilisation of ecosystem services (Ostrom, 1990). In many instances however, institutions fail to enhance the supply of ecosystem services, especially where local users have no way of organising themselves into collective groups for community self regulation in the exploitation of ecosystem services. Understanding how production and flow of ecosystem services and human well-being is mediated by institutional factors is therefore very important (Biggs et al 2015), and often requires an analysis of the political economy and cultural forces that produces and maintains these institutions. Highlighting the key roles of institutional drivers of ecosystem services and human well-being changes can help identify crucial leverage points for securing or sustaining ecosystem services and human well-being of appropriate beneficiaries.

Demographic factors: Current population structure and size coupled by primary drivers of population change such as fertility, mortality and migration affect demand for ecosystem services and influence human well-being. The rate of urbanisation also affects demand for ecosystem services and human well-being through altering the structure of consumption, but may have less impact than population size and affluence (Dietz et al., 2007, Nelson et al. 2006).

Economic growth is the main global driver of resource consumption (Dietz et al., 2007). While technological and institutional innovation have increased resource use efficiency, growth in consumption has outstripped increases in efficiency (Raudsepp-Hearne et al 2011). The distribution of economic growth, whether it improves the well-being of the poor more than the rich, also has consequences for both human well-being and the use of ecosystem services, but how these pro-poor or pro-rich economic growth reshapes relationships between ecosystem services and human well-being has not been well studied.

Sociopolitical drivers: Human conflicts are drivers of change in ecosystem services and human well-being. For instance, war-driven environmental degradation creates poverty, overexploitation of resources, underdevelopment, and, in extreme cases, famine and social destruction (Nelson et al. 2006). However, everyday conflicts over use, access, and management of ecosystem services can strongly shape the dynamics of ecosystem services, who benefits from ecosystem services, and how ecosystem services are managed (Daw et al 2011). For example, the current use of ecosystem services in the Columbia River basin in the USA is strongly shaped by historical conflicts among fishers, power users, loggers, and treaties and conflicts among colonial and Aboriginal peoples (Peterson 2000).

Cultural and religious drivers: Values, beliefs, and norms may influence decision making about the environment. Culture conditions perceptions of the world, influences what is considered important, and suggests courses of appropriate action or otherwise (Nelson et al., 2006). Guided by cultural norms, communities may be able to avert tragedy of commons through proper governance of ecosystems. Social and cultural values, such as food preferences and sacred spaces, can have a substantial impact on demand and use of ecosystem services and thus human well-being (Martın-Lopez et al., 2012).

Scientific and technological drivers: The development and diffusion of scientific knowledge and technologies have important implications for ecological systems and human well-being . Use of modern agricultural inputs improves yields and reduces pressure on land. However, overuse or inappropriate use of inputs will have adverse effects on ecosystem services (Nelson et al., 2006). Improved industrial technologies may reduce demand for natural resource based inputs.

Climate variability and change: Climate change is negatively impacting ecosystem services and consequently human well-being in many parts of the world, by adding catastrophic events such as super-storms, droughts, flooding, coral reefs and forest die-backs, sea level rise, disease outbreaks etc. Climate change will exert greater threats to the continued production of ecosystem services to meet demands for human well-being . Atmospheric concentration of carbon dioxide has increased due to fossil fuel burning and land-use changes. Atmospheric concentration of methane and nitrous oxide has also increased over time. Temperatures and precipitation patterns have also continued to change, leading to increased incidence in floods and drought, which affect supply and demand for ecosystem services (Nelson et al., 2006; Chiang et al., 2014). Accelerated sea level rise resulting from climate change is likely to affect tidal marshes and their supply of ecosystem services (Craft et al., 2008). Modelling the exact impacts of climate change on ecosystem services and human well-being

will aid mitigation and adaptation policies that could counter negative impacts of climate-driven decline in ecosystem services and human well-being (ICSU-UNESCO-UNU, 2008).

Land use changes: Four main types of land conversion affect demand and supply of ES: deforestation, dryland degradation, agricultural expansion and abandonment, and urban expansion. Deforestation is most common in the tropics, while dryland degradation is worst in drylands and hyper-arid zones of the world, mostly in Asia (Nelson et al., 2006). Land-use changes could result in increases in carbon storage, timber production, food production and decreases in habitat for certain species (Lawler et al., 2014). Changes in land use could undermine the capacity of ecosystems to sustain food production, maintain freshwater and forest resources, regulate climate and air quality, and ameliorate infectious diseases (Foley et al., 2005).

Invasive species: Invasive species can significantly modify supply and demand for ecosystem services, through ecological and evolutionary effects. Invasions by alien organisms and diseases could lead to huge losses in ecosystem services or even extinction of native species. Biological invasions could lead to losses in crops, fisheries, forestry, and grazing capacity, but in some cases, alien species (crops, trees, livestock etc) could be beneficial, especially when introduced as adaptation or mitigation against changing socio-ecological systems (Peh et al. 2015; Nelson et al., 2006).

Natural environmental disturbances: Ecosystem structure and functioning can be altered by extreme shocks such as earthquakes, and typhoons (Chiang et al., 2014) , but also by less extreme events such as droughts, floods, storms, or fires. These disturbances reshape both the supply of and demand of ecosystem services at multiple scales. The impacts of these disturbances are shaped by their interaction with ecosystems, but also anthropogenic assets, management strategies, and the human response to such disturbances. Modelling these coupled dynamics could improve resilience of ecosystem services (Biggs et al 2015), by improving ecological management strategy, ecosystem governance, disaster planning, and rebuilding strategies.

5.2.5 Scales, interactions, and feedbacks

Current conceptual frameworks such as Social-Ecological Systems and the IPBES framework emphasises the importance of the following three considerations in ecosystem services and human well-being modelling: (1) scales (e.g., local, national, regional, global, and scale transferability), (2) interactions (e.g., interactions among drivers of ecosystem services flows to beneficiaries), and (3) feedbacks (e.g., feedback loops between ecosystem services and human well-being). However, existing ecosystem services and human well-being modelling tools still need further development to capture and address above mentioned three considerations (Carpenter et al. 2009). In addition, spatial heterogeneity, stochasticity, non-linearity, and overall complexity of scales, interactions, and feedbacks need to be captured in the models of ecosystem services and human well-being to represent some resemblance to real world processes or at least gain insights on real world processes.

5.2.6 Synthesis

Modelling the linkages between ecosystem services and human well-being is much needed to address pressing issues such as biodiversity losses and the corresponding declines in ecosystem services, and their potential negative consequences on benefits that nature provides to human societies. In addition, modelling the linkages between ecosystem services and human well-being may be key to achieving international and national IPBES and CBD targets to halt biodiversity decline and to ensure equitable access to nature's benefits. The conceptual foundations for modelling ecosystem services and human well-being have advanced in the last decade; however, the empirical data and the development of models to capture the linkages between ecosystem services still needs further exploration (see Sections 5.4 and 5.5), especially in relation to context and scale (see Section 5.3). In addition, there is still a great need to expand applications of existing models of ecosystem services and human well-being across various ecosystems and scales across to globe to improve current model performance (i.e., address uncertainty) and capacities of various stakeholders in modelling ecosystem services and human well-being (see Section 5.4). Moreover, there is a great need to expand conceptual framework for modelling ecosystem services and human well-being in relation to (1) complex interactions feedback loops amongst drivers and human-ecosystem services linkages, (2) thresholds and tipping points, and (3) the importance of social-ecological contexts (see Section 5.4).

5.3 The type of assessment of ecosystem services varies with decision context

Models and scenarios can improve structured decision making by transparently representing assumptions and thinking underpinning decisions, compressing and synthesising complex information is an understandable way, and helping identify and explore new policies and unexpected outcomes. However, the value and utility of a model depends upon the decision context in which it is used.

The attributes of the decision context will determine the scope of ecosystem service modelling and scenario analysis required. A decision context can be defined by their ecological, social and decision context (Table 5.1). Important attributes of decisions contexts relevant to ecosystem services include: temporal and spatial scale, jurisdictions and administrative contexts, socio-cultural characteristics of the beneficiaries, epistemologies of the decision makers, governance and institutional settings, the decision dynamics (back-casting vs. forecasting) and the types of decisions to be made (e.g. identifying trade-offs; optimal investments; multi-criteria analyses; socio-political; experimental).

Table 5.1: Key ecological (green), social (red), and decision processes (blue) variables defining decision contexts.

Variables	*Simple*	*Complex*
Geography/ecology	Homogenous	Diverse
Flows across landscape	Weakly connected	Strong interconnections
Connection	Weakly influenced by external world	Strongly influenced by external world
Governance system	Monolithic	Poly-centric
Values	Homogenous	Conflict
Knowledge systems	Homogenous	Non-overlapping
Time period	Short term	Long term
Decision process	Unitary	Participatory
Objectives	Single objective	Multiple objective
Stakeholders	Unified	Diverse Contesting
Legitimacy	Accepted	Contested

Policy, planning and management are three different types of decision making context (see Chapter 2). Policy involves the formulation of rules and regulations to guide actions. Planning is a process of organising, prioritising and scheduling activities in order to achieve articulated goals. Management involves navigating the inevitable tensions, tradeoffs and opportunities that emerge from implementing plans and policies. Models and scenarios can improve decision making in policy, planning and management contexts by i) transparently representing key processes and assumptions that underpin decisions; ii) compressing and synthesising complex information in an understandable way; iii) helping identify unexpected outcomes, and; iv) test and explore new policies and assumptions.

There are also broader contexts in which decisions are made. There are both social contexts and ecological/biophysical contexts. There are social contexts of why a decision is being made, who is making the decision, and whether that decision maker or decision making body is considered to be legitimate. The ecological context is shaped by the properties of ecosystem services and biodiversity being decided upon, and whether the decision is a one-off decision, or part of a stream of interconnected decisions. Many decisions involving biodiversity, ecosystem services and human well-being are complex and morally fraught. Decisions may rely on poorly understood processes, that involve conflicts of interests and values among different groups in society. Structured decision

making can help improve such processes, but there is likely to be disagreement about what decision process is legitimate as well as what decisions should be made. For example, in the management of a coastal fishery, commercial fishers, indigenous groups, environmental groups and government bodies may disagree over who makes decisions, how they are made, and the boundaries of decision making. The value and utility of a model depends upon the decision context in which it is used.

A complex decision context does not necessarily require a complex model. Increase in the number of variables explicitly modelled in geometrically increases the complexity of a model, greatly increasing the difficulty of creating, parameterising, applying, analysing and communicating a model. A complex decision context complexity can often be address more simply by the application of a set of simpler models that can address complementary aspects of complexity. Alternatively, a sequential process of modelling can potential iteratively reduce the complexity of the decision context by identifying key regions, variables, and decisions, by fostering data collection and synthesis, or by building trust, and enabling communication among different stakeholders.

In this section we briefly outline the major aspects and aims of IPBES regional and sub-regional assessments followed by discussion of the likely decisions contexts for the major aspects of assessments.

5.3.1 IPBES regional and subregional assessments

The IPBES regional and subregional assessments propose to assess (IPBES/3/6), at scales yet to be determined (e.g. national, regional, river basin or other), five major aspects of biodiversity and ecosystem services:

i. Trajectories of nature's values: the values of nature's benefits to people, including interrelationships between biodiversity, ecosystem functions and benefits to society, as well as the status, trends and future dynamics of ecosystem goods and services;
ii. Trajectories of ecosystems: the status and trends of biodiversity and ecosystem services including the structural and functional diversity of ecosystems and genetic diversity;
iii. Trajectories of drivers: the status and trends of indirect and direct drivers and the interrelations of such drivers;
iv. Risks: future risks to drivers, biodiversity and ecosystems, ecosystem services and human-well-being under plausible socio-economic futures.
v. Policy responses: the effectiveness of existing responses and alternative policy and management interventions, including the Strategic Plan for Biodiversity 2011-2020 and its Aichi Biodiversity Targets and the national biodiversity strategies and action plans developed under the CBD.

The assessments will be completed for five regions (Africa, Americas, Asia-Pacific, Europe and Central Asia, and Open Oceans), with each regional assessment following a common structure but tailored to regional-specific contexts. The regional assessments will aim to answer policy relevant questions of i) the contribution of biodiversity and ecosystem services to economies, livelihoods and well-being ; ii) the status and trends of that biodiversity and ecosystem services; iii) the pressures driving change in that biodiversity and ecosystem services, and; iv) possible interventions to ensure

sustainability of the biodiversity and ecosystem services (IPBES/3/6/Add.1). The IPBES global assessment will then build on the regional and subregional assessments with processes established to ensure coherence between the two scales of assessment.

5.3.2 IPBES decision contexts

The decision contexts of IPBES are many and varied, they include the regional and national assessments, however these assessments will require many different approaches to modelling and scenario analyses. For example, the decision contexts for influencing trajectories of nature's values to humans are grounded in the social, geographical and economic sciences and will be defined primarily by the relevance of ecosystem service flows to beneficiaries. These analyses typically focus on geopolitical boundaries at scales relevant to people and shaped by available demographic data. Decisions impacting substantially on beneficiaries will likely be made at coarse scale within socio-political contexts. This will require understanding, quantifying and mapping the flows of services to beneficiaries, an area of research only recently emerging (Syrbe and Walz 2012, Reyers et al. 2013, Bagstad et al. 2014). Recent concepts for linking beneficiaries to ecosystem services include quantifying service provisioning and benefitting areas and service connecting regions. The questions asked here may include identifying natural ecosystems of high scenic beauty and recreational value and the users of these areas (Palomo et al. 2013, Palomo et al. 2014). Also of question is the location of communities most vulnerable to climate change (who will be beneficiaries of carbon sequestration and the climate regulation service) and to natural disasters such as flooding, landslides and cyclones (who will be beneficiaries of flood regulation, erosion control and extreme event moderation ecosystem services, respectively). Of note is the recent work of Renaud et al. (2013) who explore how ecosystems have an important role in reducing risks associated with natural disasters, clearly demonstrating the value of ecosystems to people. Another emerging area of research is the impact of increasing urbanisation on the demand, supply and flow of ecosystem services from agro-ecosystems, and the subsequent risks with the increased disconnect between ecosystems and people (Cumming et al. 2014).

The decision contexts for managing ecosystems are grounded in the ecological sciences. Decisions here are supported by biophysical models that aim to represent the processes that underlay the supply of ecosystem services and the changes to supply from changes in ecosystems and biodiversity. Decisions will often be location-specific and will involve identifying trade-offs in biodiversity, ecosystem and ecosystem service supply outcomes between alternative approaches to managing the land, water and biota. It is important to establish the relationships between elements of biota and physical systems and the supply of ecosystem services to provide evidence that management interventions will lead to beneficial outcomes.

The questions asked related to the trajectories of ecosystems may include understanding the efficacy of land or water management interventions for improving the condition of ecosystems and the subsequent improvements in the supply of ecosystem services. The scale of these types of decisions will generally be small (e.g. plot, paddock, river reach, vegetation community), although may extend to landscapes if ecological connectivity is of interest, and will require the collective involvement of a highly diverse group consisting of many decision makers.

Another decision context of assessments aims to understand drivers and risks to biodiversity, ecosystem services and human well-being , and the effectiveness of policy responses that mitigate risk. Decisions here will be improved by scenario analyses, potentially at a relatively coarse geographic and temporal scale and may involve any combination of trade-off analyses, optimisation, and multi-criteria analysis. For example Bryan and Crossman (2013), used high resolution spatial data, to simulate nearly 2,000 economic and biophysical scenarios to evaluate the land use changes, and subsequent impacts on the supply of ecosystem services, that may occur to the year 2050 in southern Australia following policy that creates markets for food, water, carbon and biodiversity. Using comparable methods, but for the United Kingdom, Bateman et al. (2013) explore the potential land use changes and subsequent impacts on ecosystem service supply of selected services under six plausible future socio-economic scenarios that drive land use change. There are others who have done similar work for other parts of the world, such as in the USA (Nelson et al. 2009), South Africa (Egoh et al., 2010) and Europe (Willemen et al. 2010, Willemen et al. 2012). Analyses typically forecast the impact on and trade-offs to biodiversity and ecosystem service supply and demand from external influences, such as new policy and/or climate change (Nelson et al. 2013, Bryan et al. 2014).

5.4 Types of models

5.4.1 What types of attributes differentiate ecosystem services models

Modelling the impacts on beneficiaries resulting from change in biodiversity and ecosystems takes many different forms for many different purposes (Crossman *et al.* 2013b). Models of ecosystem services tend to fall into one of two categories:

(1) simpler proxy-based models of ecosystem services underpinned by land use and land cover, and

(2) models which simulate biophysical processes and typically arrive at production functions and detailed system understanding (Kareiva *et al.* 2011; Crossman *et al.* 2013a; Maes *et al.* 2015).

Most often modelled is the supply side of ecosystem services, that is, the dynamics of the flow of services from natural capital to people. Much less common is the modelling of beneficiary demands for ecosystem services, or how changes in human populations and demographics translate to changes in demand for natural capital and the flow of services (for discussion of drivers of biodiversity and ecosystem change see Chapter 3).

In this section we summarise the key attributes of models of ecosystem service flows from the proxy to the process models, focussing on the supply side. We briefly describe the attributes, dynamics, scales, levels of complexity, and handling of uncertainty typically found in models. Different types of models can be related to different parts of the IPBES conceptual framework (Fig 5.3).

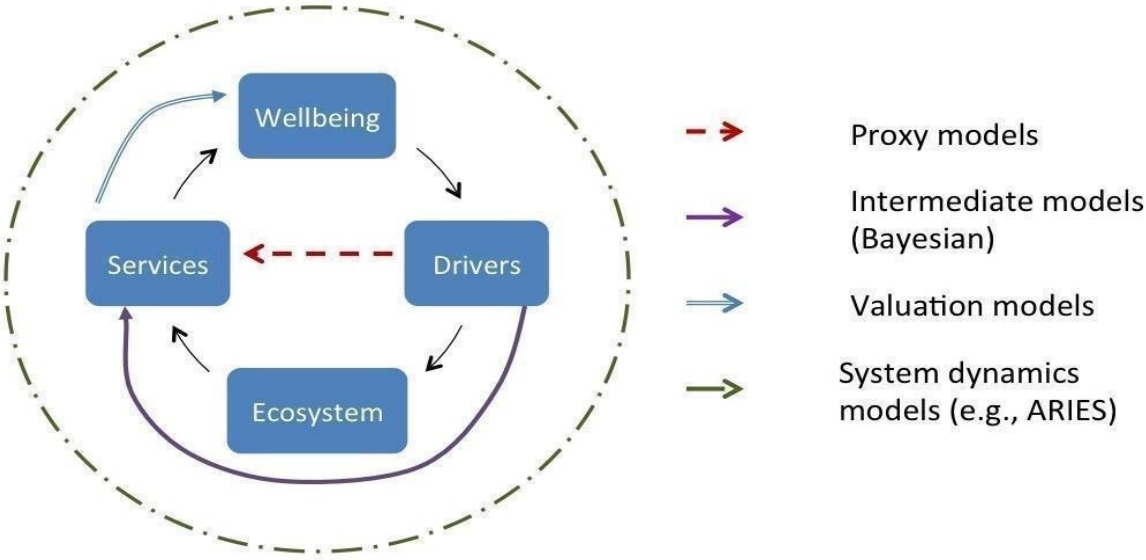


Figure 5.3.: How different types of model connect to IPBES conceptual framework. (Draft figure – will be revised based on section 5.4.3.]

5.4.1.1 Proxy-based models

At the simplest level these models are approximations of ecosystem service flows at a single point in time. Biodiversity (e.g. species distributions), land use, land cover and/or discrete elements of natural capital are usually used as proxies. For example, spatial data on perennial vegetation extent has been used to estimate the flow of ecosystem services such as moderation of extreme events (in combination with soil information, e.g. Chan *et al.* (2006), Schulp *et al.* (2012)) and carbon sequestration for climate regulation (in combination with carbon stocks, e.g. Nelson *et al.* (2009)). Soil and/or broader land cover data has also been used as proxy models for other regulating services such as soil fertility (Maes *et al.* 2012) and erosion prevention (Maes *et al.* 2012).

Ecological production functions have been suggested as a robust way to forecast the effect of human impacts on ecosystems and the supply of ecosystem services (Olander and Maltby 2014, Wong et al. 2014). According to Wong et al. (2014), ecological production functions are regression models that measure the statistical influence of marginal changes in ecosystem characteristics on final ecosystem services at a given location and time. A marginal change is the amount an output changes from an additional unit of input, all else held constant. However, such production functions will often fail when change is non-marginal, or when they are used in contexts in which key social-ecological factors are different from those in which they have been parameterised.

Simpler proxy models have improved with the addition of complexity by disaggregating land use/cover data and combining with additional information (e.g. expert knowledge, higher spatial or temporal resolution data). Although still proxy-based, these types of models better account for spatial heterogeneity and may more accurately represent ecological structures and processes. A notable study where land cover data is complemented by a number of additional datasets is the study by Schulp *et al.* (2014) who modelled the production and consumption of wild foods in Europe.

As a proxy for production, Schulp *et al.* (2014) used species distribution models to downscale coarse resolution species distribution data of important wild food species to high resolution land cover data. To model consumption, Schulp *et al.* (2014) used a mix of internet and literature searches, ingredient lists from cookbooks, and hunting statistics. A further example of the integration of expert opinion, land cover data and other empirical data is the matrix models that estimate the capacity (i.e. ability based on ecological condition and integrity) of a landscape to supply ecosystem services, pioneered by Burkhard *et al.* (2009). These models have gained popularity as a pragmatic way to quantify spatiotemporal changes in supply of multiple ecosystem services under scenarios and drivers of environmental change, especially in data sparse locations (Kaiser *et al.* 2013), and to meet co-design, participatory and transdisciplinary needs inherent in ecosystem service assessments (Fish 2011; Jacobs *et al.* 2015).

The relative simplicity of proxy models means they require fewer resources and technical expertise, making them useful where ecosystem service data is poor and measurability is difficult. Their simplicity makes them very amenable to participatory processes. Proxy models are transferable, as done in the highly influential Costanza *et al.* (1997) study and their recent follow up (Costanza *et al.* 2014) that estimated the supply and value of the world's ecosystem services across a handful of broad global biomes. But the credibility of proxy models has been questioned because of their generalisation across non-similar contexts (Eigenbrod *et al.* 2010). Typically absent in proxy methods is system dynamics such as socio-ecological feedbacks, complex interactions, temporal changes, and inclusion of external drivers of change. When these dynamics are important, or expected to play a strong role proxy models may produce inaccurate results.

5.4.1.2 Process-based models

Process based models aim to describe the ecosystem functions and biophysical processes that underlie the supply of services of benefit to people. These models estimate the flow of ecosystem services from natural capital with more realism than proxy-based methods. Process models can include socio-ecological feedbacks and interactions at fine scales, and therefore are very applicable to assess the changes to ecosystem services from changes to external drivers under a management, policy or climate scenario. Examples include the use of tree growth models, combined with stand management and spatially explicit soil and climate parameters to simulate carbon sequestration for measuring the climate regulation ecosystem service (Paul *et al.* 2013; Bryan *et al.* 2014). Hydrological process models have been used to link changes in land cover and land management to changes in the quantity of freshwater supply (Le Maitre *et al.* 2007) and the quality of freshwater (Keeler *et al.* 2012). Norton *et al.* (2012) integrated three complex process models to estimate the impact of alternative land management scenarios on freshwater quality.

Many of the process-based models of ecosystem service supply have been developed over a long time within specific scientific disciplines, such as hydrology and agronomy, that has often not been well integrated or reported in the ecosystem services literature. For example hydrologists have for decades been modelling complex hydrological processes using detail time-series climate and stream gauge data, often at daily time-steps over 100+ years, to simulate catchment scale rainfall-runoff dynamics and the outcome of interventions such as land use change or dam construction (e.g.

(CSIRO 2008)). Similarly, agronomists have built a number of crop yield simulation models using time-series climate data, soil parameters and crop management regimes which can be used to estimate the food production ecosystem service in agro-ecosystems. A prominent example is the Agricultural Production and Simulation Model (APSIM; (Keating *et al.* 2003)).

Process-based models are designed to replicate complex systems at fine scale and are often calibrated and validated against empirical time-series data to test for uncertainty. These types of models are good for sensitivity analysis through multiple model runs, and, arguably are best for testing scenarios of management, climate and policy impact. However, they suffer from needing detailed technical expertise to implement, are data and time heavy, and are not easily transferred to other locations. They may also focus on few ecosystem services, and neglect trade-offs or other interactions.

5.4.1.3 Middle ground: probabilistic models

There has been recent interest in Bayesian probabilistic models that integrate expert knowledge with multiple data sources to model flow of ecosystem services (Haines-Young 2011; Landuyt *et al.* 2013). Although not themselves models that simulate biophysical processes, Bayesian models call or take outputs from biophysical models, and then integrate with probabilistic qualitative data often derived from expert knowledge about social systems. Integrating expert and stakeholder knowledge with quantitative data makes Bayesian models very useful for evaluating scenario impacts (Keshtkar *et al.* 2013; Fletcher *et al.* 2014) in situations of limited data availability and/or where there are participatory, co-design requirements. Being probabilistic, Bayesian models explicitly account for stochastic uncertainty. Bayesian models are therefore proposed as a robust way to bridge the gap between the more accurate but less transferable and participatory process models and the simple, transferable but heavily generalised proxy models (Landuyt *et al.* 2013).

The technique of bayesian belief network have also been used to assess ecosystem services. Landuyt et al. (2013) provide a review of 47 such applications. This approach provides advantages of the ability to update and include additional data – which makes them used for applications with limited data -- and explicit treatment of stochastic uncertainty, but may be limited by the need for proprietary software, discretisation of the data, and the absence of feedback loops. Similarly fuzzy cognitive maps combine an identification of causal links with probabilistic estimations of their impact. These models can be use to make qualitative scenarios more rigorous as well as elicit decision maker models (Kok 2009).

5.4.1.4 Middle ground: social-ecological scenarios

Scenario analysis is a type of soft systems modelling that has been increasingly used to analyse the dynamics of social-ecological systems, with a strong focus on ecosystems services and human well-being (Peterson et al. 2003). Scenario analysis differs from traditional quantitative models in that they are flexible, accessible, can integrate non-quantitative, partially quantitative, or fully quantitative information (Amer et al. 2013). Social-ecological scenarios have usually analysed how decisions or policies perform across alternative futures in a way that addresses uncertainties both by improving social capacity to consider and shape the future and identify robust policies. As

frameworks for integration, scenarios provide a platform for addressing and bridging different approaches to knowledge, views of how the world works, and values (Thompson et al 2012).

Participatory scenario planning has frequently been used to address social-ecological dynamics, due to the ability of scenario planning to incorporate and engage with diverse knowledges scenarios. Prior to, but particularly since the Millennium Ecosystem assessment, a diversity of participatory social-ecological scenarios have been run in many different places around the world. These projects ranged from participatory planning around protected areas in Spain (Palomo et al 2011) and agricultural futures in central USA to evaluating investments in dryland agriculture in Tanzania (Enfors et al 2006) These projects have been used to engage diverse communities, often including indigenous people, in discussions around the management and governance of landscapes for multiple benefits. A scenario approach was used in these situations because scenarios can easily be understood as stories, can also be used for communication and outreach, enriching understanding of social-ecological dynamics, uncertainties, and options (Peterson et al 2003)

Compared to technical models, scenarios are often more accessible, integrative, and engaging; they are also better able to explicitly address tradeoffs among different groups and multiple pathways between ecological change and human well-being (Carpenter et al 2006). However, scenarios are less rigorous, less comparable, and less generalisable than technical models. A participatory scenario process can be more time consuming than a modelling exercise, but is probably comparable in time investment to participatory modelling exercises (PSES In Review). Many large assessments, as well as some smaller scenario exercises have combined quantitative models and qualitative storylines. In large assessments, this story and simulation approach (Alcalmo 2008), allows multiple complex integrated assessment models to be combined, but often runs into problems of consistency and an emphasis on quantitative results, even when non-modelled aspects of the scenarios may actually be more important, such as the dynamics of diet change or shifts in agricultural practices.

A number of guidebooks to conducting social-ecological scenario planning projects have been developed, but further improving the accessibility, diversity, and guidance on tools and techniques for scenario process management and scenario development is needed. Recent research has focussed on combining forecasting and backcasting in scenarios (Kok et al 2011), evaluating scenario methods, expanding scenarios from narratives to using different media in scenario planning (Vervoort et al 2012), as well as better use of softer quantitative modelling approaches such as fuzzy cognitive maps (Jetter & Kok 2014). However, a wider use of scenario methods requires making scenario practice more accessible, which requires building a community of practice among scenario practitioners, evaluating scenario processes, and assessing the utility of different tools for different contexts and objectives.

5.4.2 Description of major ecosystem services models

The major models (and modelling approaches) for quantifying ecosystem services are compared in Table 5.2 and then more fully explained below. The following sections describe different modelling frameworks. We have placed more emphasis on modelling frameworks that have a community of practice around them, have available documentation, and are open-access.

Table 5.2.: Summary of major ecosystem services models and modelling approaches. Dynamic models are shown in red-yellow, static or snapshot models are shown in blue-gray.

Model	Scale	Ecosystem services	Ease of use	Use in participatory processes	Reference
IMAGE	Global	7	Expert	No	Stehfest et al. 2014
EcoPath / EcoSIM	Region	3	Medium	Yes	Christensen et al. 2005
MIMES	Watershed-Global	12	Difficult	Yes	Boumans et al. 2015
LUTO	National	4	Difficult	No	Bryan et al. 2014
ARIES	Watershed or landscape	11	Difficult	Yes	Villa et al. 2014
INVEST	Watershed or landscape	17 + 9 beta	Medium	Yes	Sharp et al. 2014
LUCI	Site - watershed/landscape	7	Easy	No	Jackson et al. 2013
SOLVES	Watershed or landscape	2	Medium	Yes	Sherrouse et al. 2011
Co$ting Nature	Landscape	4	Easy	Yes	www.policysupport.org/costingnature
TESSA	Landscape	5	Easy	Yes	Peh et al. 2014
Corporate ES Review	Corporate entity	Varies	Medium	No	Hanson et al. 2012
SEEA-EEA	National	Flexible	Easy	No	European Commission et al. 2013
Matrix models	Watershed	Flexible	Easy	Yes	Burkhard et al. 2009
Green GDP/GPI	National/Regional	Flexible, but limited -> $	Easy	No	Kubiszewski et al. 2013

InVEST: Integrated Valuation of Ecosystem Services and Tradeoffs

InVest is a well-developed and widely applied suite of models for different types of ecosystem services, typically using the spatial extent and configuration habitat or land use as predictors of ecosystem services production. It has been continually developed and expanded by the Natural Capital Project since 2006 (Kareiva 2011). As of late 2014, the toolkit includes sixteen distinct InVEST models suited to terrestrial, freshwater, and marine ecosystems. InVEST models are based on production functions that define how an ecosystem's structure and function affect the flows and values of environmental services. InVEST models are spatially explicit and produce results in either biophysical terms, whether absolute quantities or relative magnitudes (e.g., tons of sediment retained or % of change in sediment retention) or economic terms, based on assumptions regarding future price and cost developments (e.g., the avoided treatment cost of the water affected by that change in sediment load).

InVEST's modular design provides an effective tool for exploring the likely outcomes of alternative management and climate scenarios and for evaluating trade-offs among sectors, services, and beneficiaries. These models are best suited for identifying spatial patterns in the provision and value of environmental services on the current landscape or under future scenarios, and trade-offs between management scenarios. With validation, these models can also provide useful estimates of the magnitude and value of services provided. Advantages of this approach are that it is transparent, open-source and freely accessible, with documentation and training available. The spatial extent of analyses is flexible, allowing users to address questions at the local, regional or global scale. The appropriate application scale is driven primarily by the quality and resolution of input data. Uncertainty in ecosystem services estimates produced by the InVEST models may be explored by performing sensitivity analyses on model inputs (e.g. Hamel and Guswa 2014). One model, carbon storage and sequestration, includes an automated uncertainty analysis in which users specify probability distributions for inputs and the model outputs include confidence intervals around carbon estimates.

Feedbacks are not explicitly built into the model structure, but are taken into account during the process of project scoping, model building, and implementation. For example, models are often applied in a context of scenario assessment, in which stakeholders explore the consequences of expected changes on natural resources using one or more of the InVEST service models. These scenarios typically include a map of future land use and land cover or, for marine contexts, a map of future coastal/marine uses and habitats, and uncertainties and feedbacks in the social-ecological system should be considered and articulated into the formulation of scenarios.

Based on 20 pilot demonstrations of InVEST in a diverse set of decision contexts, Ruckelshaus et al. (2013) have concluded that these simple production function models have been useful, with limitations appearing at the very small scale, and for specific future values. These models have been applied in multiple terrestrial, freshwater, and marine settings and in a range of decision contexts, including development and conservation planning, infrastructure permitting, climate adaptation planning, corporate sustainable sourcing, strategic environmental assessment, and designing payments for ecosystem services (PES) schemes. The application of InVEST for ecosystem services

assessment is most effective when it is embedded within an iterative science-policy process that is broadly participatory (Rosenthal et al. 2014).

InVEST models run as stand-alone software tools, but users will need a mapping software such as QGIS or ArcGIS to view results, and Python programming skills will facilitate more complex analyses such as uncertainty assessments or optimisation. Significant skill is needed to run the model: typically it will take 1-3 people two months to a year to compile data and run one or more InVEST models, although this depends on project scope and data availability. In our experience, the parts of the process requiring the most time include data collection, scenario development and iteration (i.e., re-running the models with better data and further stakeholder discussion to improve the usefulness of the model for decision-making).

InVEST provides a framework that can be adapted to the needs of specific applications. For example, Guerry et al. (2012) used the INVEST approach on the west coast of Vancouver Island in British Columbia, Canada to consider multiple services -- shellfish aquaculture harvest, spatial extent of recreational kayaking, water quality, number of recreational homes, and habitat quality -- under baseline conditions and scenarios of industry expansion and conservation zoning. They found that the conservation zoning would increase the production of all services except for the number of recreational float homes, whereas the industry expansion scenario would increase recreational float homes and shellfish aquaculture, having negative effects on habitat and water quality (Guerry et al., 2012). They used a valuation approach for shellfish harvest, but not for the other services considered, and they found that stakeholders found using different currencies for valuing different ecosystem services to be an acceptable approach.

ARIES: ARtificial Intelligence for Ecosystem Services

ARtificial Intelligence for Ecosystem Services (ARIES) is a modelling platform incorporating multi-scale process-based and probabilistic Bayesian models that has been applied in the USA, Latin America, and Africa (Villa et al., 2014). It is spatially explicit and any ecosystem services may be modelled -- ARIES focuses on final benefits, to avoid possible double-counting related to the including intermediate services. Because ARIES is accessed through a web interface, commercial GIS or modelling software is not needed. A particular advantage of this approach is the flexibility to use alternative sets of models to assess a particular system. The online Ecosystem Services Explorer demo allows users to map and quantify eight different services (carbon storage and sequestration, flood regulation, coastal flood regulation, aesthetic views and proximity, freshwater supply, sediment regulation, subsistence fisheries, and recreation) in seven case study regions. A module for nutrient regulation is under development. Initial conditions are set with a Bayesian network that feeds into non-Bayesian dynamic flow models, which include feedback. ARIES uses separate model formulations to represent source and use of a service. ARIES explicitly includes the flow of services to groups of beneficiaries, using agent-based models (Villa et al., 2013); this is significant for considering trade-offs, and for guiding policy (Bagstad et al., 2014).

Because of the model's complexity, significant time and skill are required for independent applications of ARIES, and these will likely require involvement of the ARIES development team --

new users must be registered to use the platform. Bagstad et al. (2013b) compared an ARIES to an INVEST simulation for the San Pedro river, and found that the applications took 800 and 275 hours, respectively. The model has significant data requirements, however, the ARIES system will assist users in locating appropriate datasets. The ARIES team envisions developing generalised global models available in future releases, which will make ARIES more accessible. ARIES does not include valuation, however Sherrouse et al. (2014) have used ARIES together with the Social Values for Ecosystem Services (SolVES) tool, a GIS tool to map and quantify perceived social (non-monetary) values, including biodiversity (Sherrouse et al. 2011). The SolVES tool is freely available, but requires the use of a proprietary GIS.

Ecopath with Ecosim (EwE)

Ecopath with Ecosim (EwE) was developed to dynamically represent energy flows through marine and aquatic ecosystems. EwE consists of three interlinked components: Ecopath, Ecosim, and Ecospace (Christensen and Walters 2004). First, Ecopath describes a static mass-balanced snapshot of the stocks and flows of energy (usually biomass) in an ecosystem. In typical Ecopath models, the modelled food-web is represented by functional groups that include one or multiple species with similar life history characteristics and trophic ecology and biomass removal by fishing is explicitly represented. Ecopath is described by two basic equations describing biomass production and consumption. Flows of biomass between functional groups are determined by data on diet composition. Out of 435 Ecopath models that are described in a database called EcoBase, 87% of the models were developed to answer questions regarding the functioning of the ecosystem, 64% to analyse fisheries, 34% to focus on particular species of interest, and 11% to consider environmental variability. Less than 10% of the models looked at issues related to MPA, pollution or aquaculture (Colléter et al. 2013). Second, Ecosim allows time dynamic simulation of ecosystems that are described by Ecopath. Ecosim is based on an Ecopath model to provide some of the initial-state Ecosim parameters. It uses a system of time-dependent differential equations from the baseline mass-balance Ecopath model to describe the changes in biomass and flow of biomass within the system over time, by accounting for changes in predation, consumption and fishing rates (Christensen et al., 2005; Pauly et al., 2000; Walters et al., 1997). Particularly, predator-prey interactions are controlled using an algorithm developed based on the “foraging arena theory”, through which spatial resource usages of predators and preys and their effects on their interactions are implicitly represented. It is primarily designed to explore fishing scenarios and their implications for the exploited ecosystems and fisheries catches. Ecosim also enables the representation of environmental forcing to and non-trophic interactions between functional groups. Third, EcoSpace allows spatial and time dynamic simulation of Ecopath modelled ecosystems. It allows users to explore the effects of spatial fisheries management policies such as Marine Protected Areas.

Ecopath with Ecosim (EwE) has been widely used to generate scenarios of changes or management of fishing effort on flows of ecosystem services from marine ecosystems through fishing. For example, EwE modelling was applied to explore the implications of limitation of beach seine fisheries on the well-being of coastal communities in Mombasa, Kenya, with a particularly focus on the poor group (Daw et al 2015). Specifically, EwE provided expected ecological and fisheries responses of the Mombasa coral reef and seagrass ecosystem under a range fishing effort scenarios. The model

represented trophic interactions between 56 functional groups and the effects of five fishing gears, including beach seine, fish trap, spear, hook and line, and net. Simulations were run to explore changes in fishing effort of these gear groups from now to 2030. Total catch, net present value of catches less fishing costs (discount rate = 11%) and total fish biomass were collected to provide indicators of food production, profitability and conservation as well as catch per unit effort by gear and by functional group. Functional groups were classified as high or low value based on fisheries monitoring data. The outputs from the EwE models were used to explore human well-being implications by using a rule-based 'toy' model to combine the key linkages of fish abundances, catches with well-being of individual stakeholders. The 'toy' model was used in a participatory workshop in which groups of stakeholders in the region was asked to explore ways to manage fishing effort of different gear groups that would maximise the well-being of specific fishing gear groups or seafood traders (Daw et al. 2015).

IMAGE 3.0

IMAGE 3.0 is integrated assessment modelling framework has been developed to analyse the dynamics of how global, long-term environmental change and sustainability problems (Stehfest et al, 2014). IMAGE contains an ecosystem service module that quantifies the supply of eight ecosystem services, using other components in the IMAGE 3.0 framework, and where necessary combined with relationships between environmental variables and ecosystem services supply, derived from literature reviews (Schulp et al., 2012). Ecosystem services derived directly from other IMAGE components include the food provision from agricultural systems; water availability; carbon sequestration; and flood protection. Estimation of the services, wild food provision, erosion risk reduction, pollination, pest control and attractiveness for nature-based tourism, requires additional environmental variables and relationships (Maes et al., 2012; Schulp et al., 2012), in particular fine-scale land use intensity data from the GLOBIO model (Alkemade et al 2009). IMAGE compares the supply of different services to the estimates of the minimum quantity required by people to to assess surpluses and deficiencies. This translate, for example, into minimum amounts of food and water for humans to stay healthy, or the minimum amount of natural elements in a landscape to potentially pollinate all crops. The fraction of people or land sufficiently supplied by ecosystem services is derived at different scale levels (Kok et al., 2014, CBD TS 79: http://www.cbd.int/ts/default.shtml)

Integrated Systems Dynamics Models

Integrated system dynamics models have also been used to translate biodiversity and ecosystem properties to ecosystem services and benefits, within the context of large-scale feedbacks between natural capital and human made capital. The earliest of these models was the Global Unified Metamodel of the Biosphere (GUMBO) model (Boumans et al., 2002). THE GUMBO model was used by Arbault et al. (2014) to consider life cycle analysis. The Multiscale Integrated Earth Systems Model (MIMES) builds on the GUMBO model using a spatially explicit approach and valuation methods for most ecosystem services (Boumans and Costanza, 2008). MIMES has been developed in Simile software, a commercial software package. Similarly, Fiksel et al. (2013) have used a systems approach to consider linkages among economy, society, and environment, where flows of ecosystem services provide value to both the economy and society. This approach has been

implemented in the VENSIM software, which requires purchase of a license for commercial or government use. An advantage of systems dynamics models is that they are comprehensive, and represent feedbacks among sectors within each time-step. However, their complexity and lack of clear documentation limits the usability by a wider group of users.

Matrix Models

Combining maps of land cover and land covers contribution to ecosystem services using GIS and matrices allows simple and rapid exploratory ecosystem service assessment that does not require access to or training in other ecosystem service assessment models such as EcoPath/EcoSim or Invest (Jacobs et al. 2015). For example, O'Farrell et al (2012) scored urban ecosystem services from remnant native vegetation in Cape Town, in which ecosystem services production by different land use classes was estimated based on expert opinion. They then analysed historical and potential future land cover change, as well as diffuse spatial benefits away from remnant vegetation. This approach allows relative cheap rapid identification of key areas and issues, and therefore enables useful discussions between ecosystem service experts and urban managers, but it does allow analysis of changes in land management or differences among particular sites (O'Farrell et al 2012). Similar approaches have been used in many other GIS oriented analyses.

Other Ecosystem Service Toolkits

There are several additional proxy-based or screening level tools and approaches available from a variety of sources that relate ecosystem state to ecosystem services (but do not include valuation). The Corporate Ecosystem Services Review (Hanson et al., 2012), developed by the World Resources Institute, is a structured methodology that helps businesses that interact with ecosystems to connect ecosystem health to business risks and opportunities. The Ecosystem Services Review uses a qualitative approach to consider the 27 ecosystem services given in the Millennium Ecosystem Assessment. TESSA is a toolkit that uses decision trees to guide users, through a process to rapidly prioritise ecosystem services for assessment, identify data needs, and communication approaches. It provides a template that users must adapt to specific cases (Phe et al 2013, Birch et al 2014). Co$ting Nature (Mulligan et al., 2010) models changes in four ecosystem services (carbon storage, water yield, nature-based tourism, and natural hazard mitigation) under scenarios of climate or land-use change. Similarly, the Land Utilization & Capability Indicator (LUCI) is a framework that considers services of production, carbon, flooding, erosion, sediment delivery, water quality, and habitat, based on GIS land and soil information (Jackson et al, 2013). These three approaches, ESR, Co$ting Nature, and LUCI, are compared in Bagstad et al. (2013).

Green Accounting Approaches and Related Indicators

There are a number of accounting frameworks that propose extending conventional economic accounts (e.g. those from the System of National Accounts –SNA) to the environment as in the System of Environmental-Economic Accounting (SEEA) and the SEEA Experimental Ecosystem Accounting (SEEA-EEA). These national/regional scale accounting frameworks seek to factor in the contribution of nature to the economy with the purpose of measuring economic performance, as well as the interaction between economic and environmental systems. These frameworks

additionally recognise that these accounting systems are important for other applications, as for example the analysis of productivity or well-being.

This consideration and extension of the economic accounts toward the environment are usually referred as green accounting (Smulders 2008). Several indicators result from green accounting frameworks in both physical and monetary units. Some proposals consist in extending the conventional GDP metric to those non-market outputs resulting from nature (often called green GDP)(Boyd and Banzhaf 2007). These authors additionally develop an ecosystem accounting framework consistent with the System of National Accounts for the treatment of nature's outputs that are used as intermediate inputs for the production of a final good, as these nature's contributions need to be additionally combined with anthropogenic assets for the co-production of final goods. Moreover, when aggregated monetary measures of economic performance are adjusted by the changes (often depletion) of environmental assets, to the result is economic metrics which are ecologically adjusted by natural capital depletion. An example is the Adjusted Net National Product (Barbier, 2012). Another metric of economic welfare is (inclusive or comprehensive) wealth accounting, where natural capital is accounted for along with measures of produced, human and social capital. The key idea behind wealth accounting is that the future consumption possibilities (including non-market benefits such as ES) depend upon the various capital types (or asset base) of a nation (Arrow et al. 2012, Dasgupta 2009, UNU-IHDP and UNEP 2014, World Bank 2011).

There are in addition other yardsticks that aim at measuring welfare by factoring also in those changes in the environment that affect human well-being as it is the case of the Index of Sustainable Economic Welfare (Daly and Cobb 1989) and the Genuine Progress Indicator (Kubiszewski et al 2013). Composite indices of welfare have also sought to combine different constituents of well-being into a single value to represent advances in human development. An example of this type of indices is the Human Sustainable Development Index (Togtokh and Gaffney, 2010), which combines the human development index with carbon emission to approximate the sustainability of human well-being. Other examples include OECD's better life index (OECD 2015).

Green accounting is also used to support decisions regarding the evaluation of different alternative scenarios by means of social cost-benefit analysis. In this context, TEEB has developed or is developing case-specific frameworks for bringing ecosystem services into economic decision making, such as for water and wetland management (Russi et al 2013), agricultural systems, and oceans (TEEB 2013).

Despite the several efforts to understand progress in human welfare by accounting also for the environment, most of these accounting frameworks fall short in capturing all aspects of human well-being . These research areas are rapidly changing as researchers attempt to better define practical definitions and measures of human well-being , however different approaches provide complementary insights.

5.4.3 Comparing model types across decision contexts

Existing models of ecosystem services are primarily focussed on the landscape scale, but there are a number of complex large-scale oriented models (Fig 5.4). However, there is a lack of large scale, non-global models.

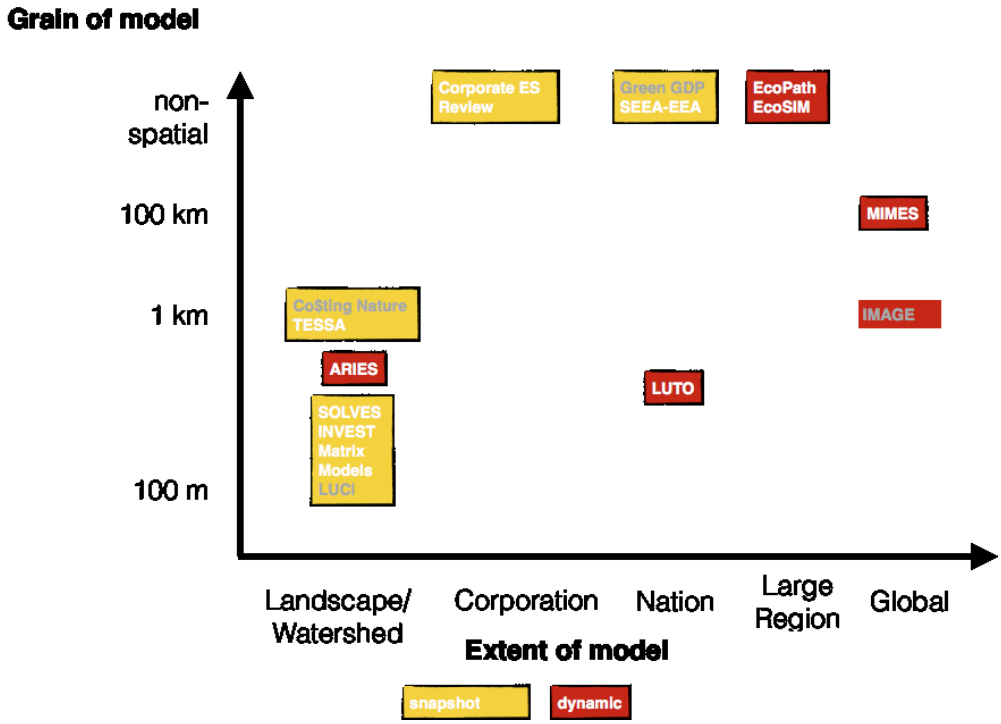


Figure 5.4.: Rough guide to spatial grain and extent of different models, along with whether they are dynamic or snapshot.

Only two ecosystem service frameworks have substantial communities of practice: InVest and Ecopath/Ecosim (Figure 5.5). Both these frameworks are moderately complex to use, and have quite different strengths and weaknesses. EcoSim focuses on fishing related ecosystem services, primarily non-spatially, but with a strong focus on dynamics and different beneficiaries. Invest is more static and is a set of interrelated models that assess individual ecosystem services. It is therefore broader, but Invest is not dynamic and is not easily linked to multiple beneficiaries. There appear to be substantial opportunities for other model frameworks, especially the easy to use ones, to build larger communities of practice.

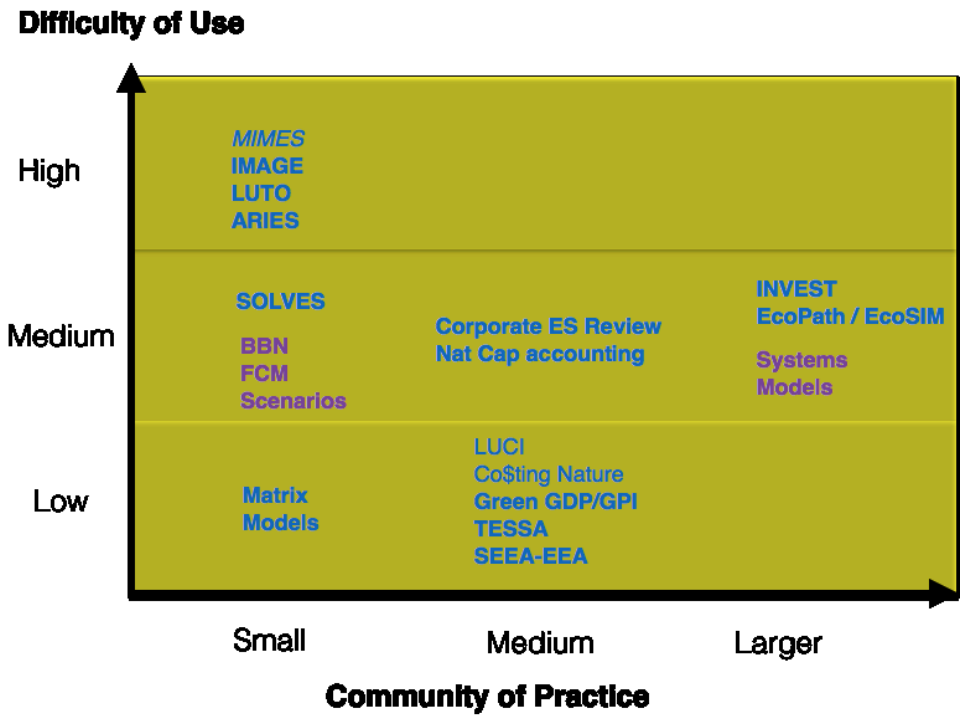


Figure 5.5.: Comparison the difficulty of use and community of practice existing for different modelling frameworks and approaches. Models with limited documentation are shown in italics (no documentation) and partial (plain text). Modelling approaches are shown in purple, frameworks in blue.

There is substantial diversity among models in how they conceptualise ecosystem services in terms of whether they focus on supply (or ecosystem service potential), or the realised demand for ecosystem services. There is also divergence between models that quantify ecosystem services or attempt some sort of valuation, and whether they link benefits to specific beneficiaries (Figure 5.6).

A key strength of modelling frameworks is if they can be adapted to new contexts. Furthermore, in decision contexts in which a problem is unclear or contested, modelling often needs to be done in a participatory fashion. Roughly half of the modelling frameworks can be used in a participatory fashion, and nearly half of them can be adapted to be used in new participatory contexts (Figure 5.7).

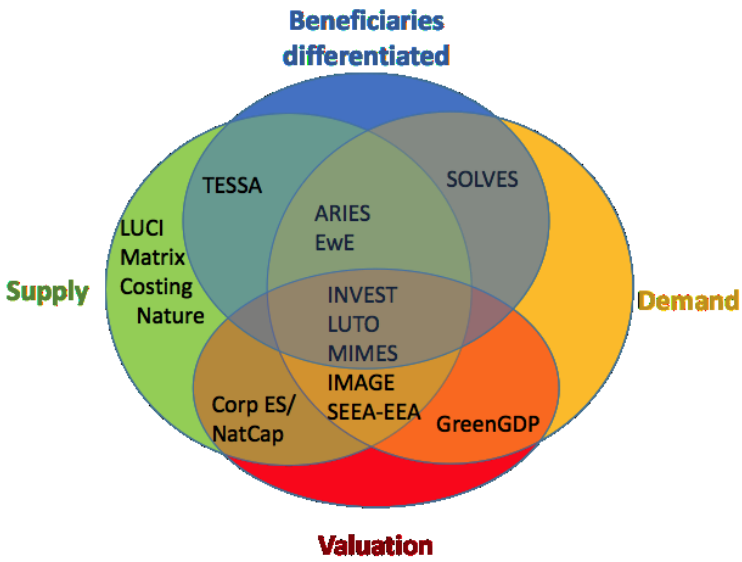


Figure 5.6.: Comparing modelling approaches and frameworks on a) the way they address ecosystem services in terms of ecosystem service supply and demand, as well as whether they include explicit valuation and have the potential to differentiate beneficiaries.

	Non-participatory	Participatory
Non-adaptive	**IMAGE** SEEA-EEA Green GDP CorpES/NatCap	**MIMES**
Adaptive	LUCI **LUTO**	**ARIES** **INVEST** **Matrix** **Co$ting Nature** EwE SOLVES TESSA

Figure 5.7.: Ability of modelling frameworks to be adapted to new systems and used in a participatory fashion.

Depending on the needs of a decision context different modelling frameworks are better suited to the task. Below we present a preliminary decision tree that defines which modelling framework fits a task based on the scale, time perspective, type of ecosystem service, and whether participation is desired (Figure 5.8).

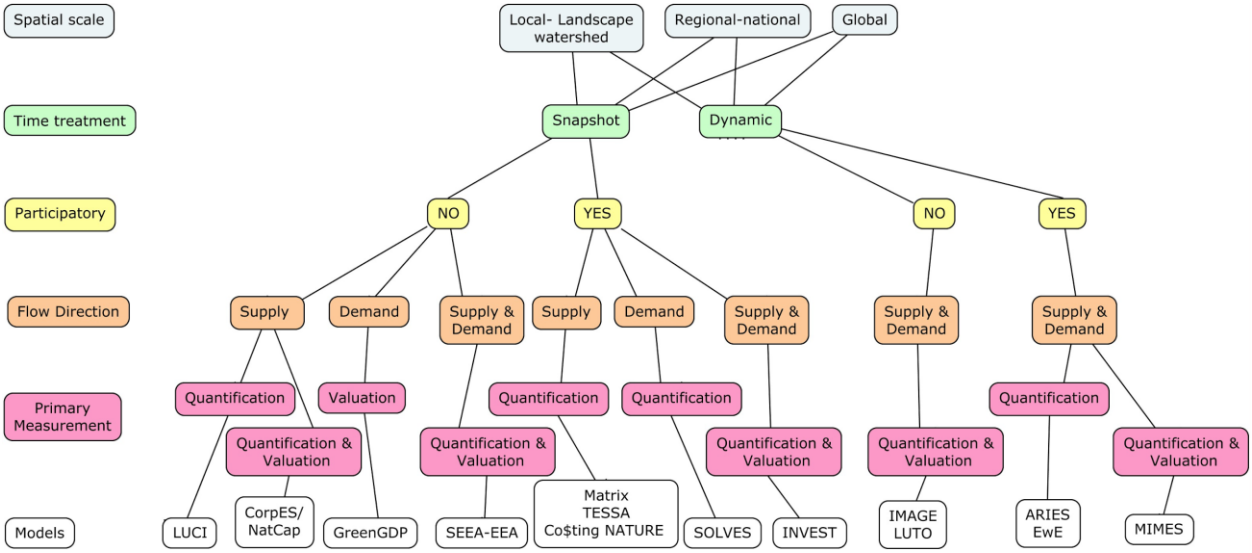


Figure 5.8 [draft]: Decision tree - showing different variables and how they can lead to different models.

Of the existing set of ecosystem service models:

- Snapshot/proxy models tend to be easier to use and have larger user communities
- Half of the models consider both supply and demand, and a majority of these have beneficiaries differentiated
- Half of the models are multi-objective, and half of the models are both participatory and adaptive, but these surprisingly these subsets of models do not completely coincide

To date, most ecosystem model development has focused on assessing ecosystem services, with minimal attention being focused on how these ecosystem services link to human well-being , especially of diverse groups of beneficiaries. Modelling the impact of ecological changes on human well-being is not well developed. This is partially because understanding of human well-being is poor, but also due the lack of involvement of human wellbeing researchers in ecosystem service modelling.

Developing tools that better link human well-being and ecosystem services will require investment and trans-disciplinary collaboration of policy makers, with natural and social scientists to develop new frameworks, methods, and tools. Most modelling tools have been developed with government decision makers in mind, and there is a need for tools that are aimed to be used and adapted by other actors, and especially that can be adapted to fit with other knowledge systems. Particular issues that need more model development are: impact of ecological change on different groups of people, incorporation of different knowledge systems in modelling operation and practice, consideration of co-production and disservices, spatial distribution of services and beneficiaries, and adapting model communication for different decision contexts.

5.4.4 Data needs for model development, and for ongoing evaluation and calibration

Spatial data describing, ideally at high spatial and temporal resolution, the following:

- Land use, land cover, and land management
- Species presence and absence
- Physical attributes (e.g. soil, water, topography, geology, climate)
- Ecosystems (e.g. vegetation communities; biomes; primary production)
- Geo-political boundaries (e.g. country/state/local governments)
- Built infrastructure (e.g. cities and built-up areas; roads; dams)
- Protected areas and conservation zones (parks; green space; camping and recreational features)
- Demographic data (population characteristics)
- Socio-economic data (land values, agricultural production and value; fish capture data, non-monetary values, indicators of well-being)

The above is primary data, but where possible if spatially explicit ecosystem service data is available, then even better. For example, the MAES project under EU FP7 has mapped many ecosystem services across the EU member states; the US EPA has a similar effort with their EnviroAtlas.

5.4.5 Knowledge needs for model development, and for ongoing evaluation and calibration

[here present evidence to support 5.7.2 research frontiers]
[need to finish 5.4 first]
[link to introduction section]
[Need to link to other knowledge systems here]
[need to link to computer science & simulation literature on model quality - salience, legitimacy, etc]

5.5 Methods for assessing, and communicating, uncertainty

[Note: section needs to match general discussion of uncertainty in IPBES, but received too late to incorporate]

Decision making is shaped by two types of uncertainty: decision uncertainty that is uncertainty about appropriate course of action; and information uncertainty that is lack of information or knowledge about how the world works (Bark et al. 2013). Information uncertainty can be either statistical or scientific. Statistical uncertainty (e.g., climate variability) is the natural variation in a system which can be quantified through probability distributions, which can then be used to derive confidence intervals and risk profiles. Scientific uncertainty describes the lack of complete knowledge in a modelled system and its parameters and is a typical feature of complex socio-ecological systems that contain non-linear relationships, unpredictable stochastic behaviour, and unknown system conditions. Although scientific uncertainty cannot be quantified, it can be reduced

to statistical uncertainty by collecting more data or improving system understanding. Scientists recognise that both “known unknowns” and “unknowable” uncertainties exist.

Bark et al. (2013) make it clear that in ecosystem service assessments, much statistical and scientific uncertainty exists because of the complex physical and ecological systems that underpin supply of ecosystem services, plus the large uncertainty inherent in socio-economic systems that value/demand ecosystem services. Schulp et al. (2014) document five sources of uncertainty in ecosystem services quantified across Europe. They identify uncertainty in the i) indicators, definitions and framework that classify ecosystem services; ii) level of process understanding which leads to different quantification methods; iii) purposes of quantification that influences the selection of indicators; iv) biophysical and socio-economic input data, and v) models used to quantify ecosystem services.

Robust communication of uncertainty has long occupied the IPCC. For the 5th Assessment Report (Mastrandrea et al. 2010), the IPCC used an elegant system that qualitatively describes the levels of confidence in reported findings, based on expert judgement, determined through evaluation of evidence and model agreement. They also used quantitative reporting of uncertainty that stems from statistical or modelling analyses, expert opinion, or other quantitative analyses. They describe uncertainty using a likelihood scale to express a probabilistic estimate of the occurrence of a single event or outcome.

5.6 Capacity Building needs

There is substantial need of capacity building in modelling and scenarios for ecosystem services, especially in the developing world. There is substantial overlap between the capacities required for modelling and producing scenarios, as scenarios frequently incorporate models and model outputs (see Linking chapter). There is a need to build communities of practice around specific modelling and scenario tools, as well as interconnections among the entire community. The natural capital project, EwE, TEEB, and ESP provide examples of communities of practice built around ecosystem services modelling and scenarios. They provide guidance in what can effectively allow models to move from academic papers to useful tools. These factors include investment in training and networking, and access to models and data.

5.6.1 Training and networking:

The continued use of models requires the development of a community of people who know how to use the model. While enabling people to use a model requires both good models and good documentation, it also requires effective, accessible training. Successful modelling frameworks have provided training through creating course material and textbooks, and also linking model training to scientific societies, existing workshops or training programmes, school curriculum, or profession certification. Training efforts often require substantial investment of time and money before resulting in a substantial community of practice, but experience shows that such efforts are essential

to enable widespread use of tools (EwE). The availability of open source training materials and example model applications allows users to train themselves, and to provide training to others, which lowers the cost of training efforts.

There is a substantial lack of models that view ecosystem services from more cultural perspectives, or that include provisioning services that are locally or regionally important, but globally marginal (e.g.,rattan, kola nut). Most models are written in programming languages based in english, and data and training are often only available in English or other major global languages, rather than indigenous or other languages. These facts suggest that there should be emphasis of developing modelling frameworks to address indigenous perspectives on ecosystem services in ways that can be generalised across multiple contexts. Such work, especially if it aids in translating, synthesising and comparing multiple perspectives could be particularly useful for minority and indigenous communities engaging in co-management or negotiation with states (Tengö et al 2014).

5.6.2 Access to models and data:

When models -- and the data needed to support these models -- are readily and easily accessible, this improves modelling capacity, by decreasing the difficulty of creating a modelling project and increasing the quality of models by allowing existing models to be evaluated. For models to be widely used they should be available at no cost. Even minimal cost can prevent people from trying a modelling framework to determine if they are useful. Some modelling tools, such as Vensim, offer version that are free for academic use or free for a period of time or with limited functionality to allow people to begin to use these tools. Other more technical tools, such as R are open access. To ensure that people not working for rich organisations have access to models it useful to develop models using open access or free tools, and to develop models that are themselves open access and free. Ecosystem service models should also use an open-source approach, where the model code is available. Additionally, these models should use good modelling practice, such as using standardised model description, and ensure that there is clear and accessible documentation that supports the model.

Databases of models that are available for download, especially if such models are linked to research products and clear documentation, can greatly accelerate the adaptation of modelling frameworks. For example, the openABM project (openABM.org) provides an excellent example of a general model database. Some journal require that ABM papers are required to submit a version of their model with documentation to the openABM database, which makes it easier for others to learn about a model, test it, or adapt it. Such practices should be encouraged by IPBES.

Data needs are widely shared across modelling frameworks, consequently, free open access to data needed to define and drive ecosystem service models and scenarios is essential to increase the ability of people to create such scenarios. If data is available in formats that suit particular model formats, that further lowers the cost. In the global change community projects such as CIESIN have assembled multiple datasets to make it easier for modeller to find data. Such data portals are currently provided by UN and other organisations (e.g. FAO, WRI), but further improving the accessibility, interconnection, and metadata of data related to ecosystem service models and

scenarios (see section on drivers) increase the ease with which models can be created. However, some models require specific types of data, for example Ecopath with Ecosim requires food web data. Two additional factors are important here - first is that currently data is often focussed on rich countries and the global scale. Because local and regional models often have different data requirements, it their development could be enhanced if databases were developed to support IPBES regional assessments, especially data that is useful for assessing ecosystem services from Indigenous and local perspectives which may not be otherwise accessible. Second, current models focus primarily on provisioning ecosystem services, carbon and water related regulating services, and tourism or other recreation related cultural services. Idiosyncratic provisioning services, non-water or carbon related regulating services and most cultural services are neglected. This gap is especially important for indigenous and local knowledge.

5.7 Synthesis & Research Frontiers

The following areas are key gaps and potential areas for rapid progress:

- Develop models for additional services, in particular non-water or carbon related regulating services and most cultural services (including those that are regionally or culturally important)

- Increase availability of data for modelling in developing regions, at regional scale, develop libraries of models, and semi-automate model set-up

- Refine tools/models/frameworks that consider multiple objectives, and enable exploration of tradeoffs and synergies. Martinez-Harms et al (2015) note that assessment of tradeoffs and prioritising of management actions are necessary steps in decision-making for ecosystem services.

- Develop tools that better link changes in biodiversity and ecosystems, to ecosystem services, and human well-being of diverse beneficiaries. There are cases in which this has been done (e.g. Daw et al 2015), but these modelling efforts are complex and focus on few ecosystem services.

- Develop theory and models of interaction between use, supply and demand for ecosystem services. This is especially important for understanding rapid and non-marginal change

- Develop better frameworks for linking multiple models and better linking models and qualitative narratives in scenario planning

- Assess how to best develop communities of practice around ecosystem service modelling (e.g., Laniak et al., 2013).

- More testing and cross-validation of ecosystem service models

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[references are neither consistently formatted nor complete due to lack of shared reference management software among authors - we are working on this & will be fixed in next version of paper]

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