# Individual chapters of the methodological assessment of scenarios and models of biodiversity and ecosystem services (deliverable 3 (c))

The fourth session of the Plenary in its decision IPBES-4/1 approved the summary for policymakers (SPM) of the methodological assessment report on scenarios and models of biodiversity and ecosystem services and accepted the individual chapters (IPBES/4/INF/3), based on the understanding that these chapters would be updated to reflect the SPM as approved.

This document, posted on the IPBES web site on 31 August 2016, presents the final individual chapters of the scenario assessment, which reflect the SPM as approved.

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# 1 Overview and vision

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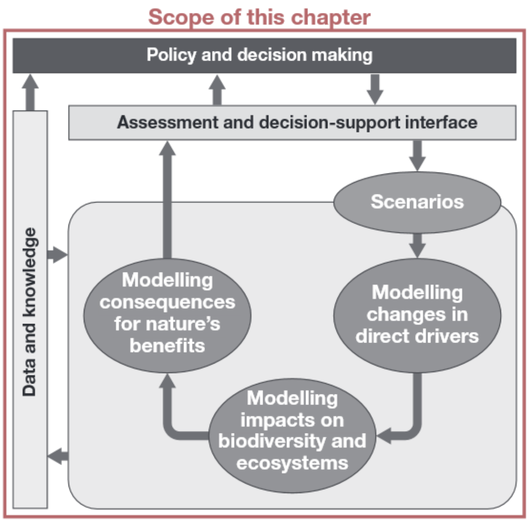
**Purpose of this chapter:** Introduces the background, purpose and scope of the Methodological Assessment of Scenarios and Models of Biodiversity and Ecosystem Services; provides a general introduction to the role of scenarios and models in policy and decision making; and outlines the structure of the remaining chapters of the report.

**Target audience:** A broader, less technical audience than for the other chapters of the report, each of which examines in greater depth a subset of issues and challenges associated with scenario analysis and modelling. Readers interested in obtaining only a general overview of the topic of scenarios and models may wish to read no more than this chapter.




## 1.1 Introduction

For the purposes of this assessment, 'models' are defined as qualitative or quantitative representations of key components of a system and of relationships between these components.

Throughout this assessment, and in most Intergovernmental Platform on Biodiversity and Ecosystem Services (IPBES) activities, the term 'models' usually, but not exclusively, refers to quantitative descriptions of relationships i) between indirect drivers and direct drivers, ii) between direct drivers and nature (including biodiversity and ecosystems), and iii) between nature and nature's benefits to people (including ecosystem services). Each of these relationships is discussed in more detail later in this chapter.

In this assessment, 'scenarios' are defined as plausible representations of possible futures for one or more components of a system, or as alternative policy or management options intended to alter the future state of these components.

Throughout this assessment, the term 'scenarios' usually refers to plausible futures for indirect or direct drivers, or to policy interventions targeting these drivers. The consequences of these scenarios for nature and nature's benefits to people are then typically evaluated using models as defined above.

Scenarios and models have the potential to contribute significantly to achieving the overarching goal of IPBES 'to strengthen the science-policy interface for biodiversity and ecosystem services for the conservation and sustainable use of biodiversity, long-term human well-being and sustainable development'. Their use in assessments, policy support and decision making offers many benefits, including to 'better understand and synthesise a broad range of observations; alert decision makers to undesirable future impacts of global changes such as land-use change, invasive alien species, overexploitation, climate change and pollution; provide decision support for developing adaptive management strategies; and explore the implications of alternative social-ecological development pathways and policy options. One of the key objectives in using scenarios and models is to move away from the current reactive mode of decision making in which society responds to the degradation of biodiversity and ecosystem services in an uncoordinated, piecemeal fashion to a proactive mode in which society anticipates change and thereby minimises adverse impacts and capitalises on important opportunities through thoughtful adaptation and mitigation strategies' (IPBES/2/17, annex VI[1]).

### 1.1.1 Purpose and scope of this assessment

The Methodological Assessment of Scenarios and Models was initiated to 'establish the foundations for the use of scenarios and models in activities under the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services' (IPBES/2/17, annex VI, www.ibpes.net).

It is one of the first assessment activities of IPBES because it provides guidance on the use of scenarios and models in regional, global and thematic assessments, provides IPBES task forces and expert groups with recommendations in terms of supporting and mobilising scenarios and modelling expertise, and identifies key gaps that need to be addressed in collaboration with the scientific community, policymakers and others. There are a large number of reviews providing typologies of scenarios and models and summarising their strengths and weaknesses (Coreau et al., 2009; IEEP et al., 2009; Bellard et al., 2012; Kelly et al., 2013; Harfoot et al., 2014a; Rounsevell et al., 2014), but all of these have a much narrower scope than this assessment and do not provide recommendations that are specifically adapted to the IPBES mandate. Overall, this assessment provides an overview of scenarios and models, a critical analysis of the types and uses of scenarios and models currently available, and perspectives on the development of new methods in the near future.

There are several audiences for this methodological assessment, with the primary audiences differing substantially between the Summary for Policy Makers (SPM), Chapter 1 and the following chapters.

The SPM and Chapter 1 have been written with non-experts in mind so that they are accessible to a broad audience, including members of the IPBES plenary, policymakers and other stakeholders. The critical analysis and perspectives in Chapters 2-8 of this assessment are more technical in nature and address the broader scientific community in addition to the expert groups and task forces of IPBES. In all of the chapters, highly technical descriptions and jargon have been kept to a minimum.

The intended target audiences within IPBES include:

- Plenary, Bureau and Multidisciplinary Expert Panel: the SPM and Chapter 1 provide a broad overview of the potential benefits and caveats in making use of scenarios and models, better integration across existing IPBES activities and priorities for future activities of IPBES;
- Task forces and expert groups involved in catalysing, facilitating and supporting the use of scenarios and models within IPBES and beyond: the full assessment provides guidance on priorities and proposed solutions for linking work on scenarios and models across IPBES deliverables, and for mobilising the broader scientific community;
- Regional, global and thematic assessments: the SPM and Chapter 1 give all involved experts an overview of the benefits and caveats in making use of scenarios and models, and provide experts working specifically on scenarios and models with guidance on more technical issues related to the application of scenarios and models in assessments.

Target audiences outside of IPBES:

- This document provides guidance to policymakers and implementers at local to global scales, as well as assessment and decision-support practitioners employing scenarios and models. Guidance to these audiences focuses on the appropriate and effective use of scenarios and models across a

[1] For official IPBES documents cited in this assessment, see the IPBES website at www.ibpes.net under the tab 'Plenary Sessions'. The first number in the IPBES document reference indicates the number of the plenary session.

broad range of decision contexts and scales.

- For the scientific community and science funding agencies: this assessment provides analyses of key knowledge gaps that, if filled, would greatly increase the utility of scenarios and models for IPBES and other science-policy interfaces. Summaries of these knowledge gaps can be found in the SPM and Chapter 1, with more detailed analyses in subsequent chapters, especially Chapter 8.

The scope of the assessment covers a broad range of scenarios and models. The objective is to provide guidance for 'evaluating alternative policy options using scenarios and models; including multiple drivers in assessments of future impacts; … including input from stakeholders at various levels; implementing capacity-building mechanisms to promote the development, use and interpretation of scenarios and models by a wide range of policymakers and stakeholders; and communicating outcomes of scenario and model analyses to policymakers and other stakeholders' (IPBES/2/16/Add.4, www.ibpes.net).

Follow-up work by an expert group is envisaged to start following the completion of this assessment in 2015 and will continue through 2017 and possibly beyond. One of the tasks of this expert group will be to establish an 'evolving guide' on scenarios and models.

The exact nature of this evolving guide remains to be defined but, since methods are changing very rapidly, it is important that the guidance provided in this assessment is updated on a regular basis. The expert group will also interact with other IPBES deliverables and the broader scientific community to stimulate work on scenarios and models that support IPBES objectives. It is envisaged that this will be similar to the interactions between the Intergovernmental Panel on Climate Change (IPCC) and the scientific community that have been created to develop scenarios and models for climate change assessment.

### 1.1.2 Background and context

Scenarios and models of biodiversity and ecosystems have been a key component of most global, regional and national environmental assessments carried out over the last decade.

The IPCC, which is the institutional equivalent of IPBES for climate change issues, has amply demonstrated the power of scenarios and models as a cornerstone of the science-policy dialogue surrounding climate change and in popularising climate change issues. The use of scenarios and models of biodiversity and ecosystem services in global and sub-global assessments is more recent. The first global assessment with a substantial component of scenarios coupled with models of biodiversity impacts was the Millennium Ecosystem Assessment (MA) released in 2005 (MA 2005). Assessments with significant use of scenarios and models to evaluate ecosystem services are even more recent (e.g. UK NEA, 2011).

Scenarios and models in assessments of biodiversity and ecosystem services have played an important role in agenda setting by alerting the scientific community, natural resource managers and politicians to the possible future risks for biodiversity and ecosystem services, and to some extent in policy formulation by illustrating possible solutions for reducing these risks (Wilson et al., 2014). Examples include the most recent IPCC assessment and the MA which have called attention to the possibility of a greatly increased species extinction risk by 2050 driven by future land-use and climate change (MA, 2005; IPCC, 2014a). The most recent Global Biodiversity Outlook used scenarios and models to call attention to the transformations of socio-economic development paths that are needed to achieve internationally agreed upon goals for climate, biodiversity and human development by 2050 (sCBD, 2014; Leadley et al., 2014; see also Table 1.1 in Section 1.5.2).

Scenarios and models of biodiversity and ecosystems are used in many contexts outside of global, regional and national environmental assessments. In particular, a wide range of policy support methodologies have been developed to allow the more direct use of scenarios and models in policy design, implementation and evaluation (see Chapter 2). The bulk of this work has been done at local scales (see two examples in Table 1.1), but some methodologies are also pertinent at national to global scales. Experience shows that the successful application of models and scenarios to policy design, implementation and evaluation requires sustained interactions between stakeholders, managers, policymakers and modellers. Numerous examples illustrating these applications are provided in this and subsequent chapters, particularly in boxes describing case studies.

A variety of approaches have been used for developing and presenting scenarios and models in environmental assessments, and very rapid progress in the development and use of scenarios and models of biodiversity and ecosystem services over the last decade (Figure 1.1) means that IPBES is now well positioned to make substantial use of these methodologies in all of its activities.

In some cases, assessment bodies have opted to support the development of a common set of scenarios of direct and indirect drivers, as well as accompanying models of impacts on biodiversity and ecosystems. Examples include the global assessments such as the Millennium Ecosystem Assessment (MA, 2005), early Global Biodiversity Outlooks (sCBD, 2006), and Global Environment Outlooks (e.g. UNEP, 2007), as well as some national and regional assessments (Southern Africa, van Jaarsveld et al. (2005); Japan, SSA (2010; UK, UK NEA, 2011; Wilson et al., 2014). At the opposite end of the spectrum, some assessments have focused on synthesising a broad range of published analyses of scenarios and modelling studies available in the literature (e.g. sCBD, 2010; Leadley et al., 2010; UNEP, 2012). Still others fall in between these extremes: for example, IPCC climate modelling has traditionally relied on a common set of scenarios of direct and indirect drivers developed specifically for the assessment, while assessment of projected impacts on biodiversity and ecosystems is primarily based on analyses of peer-reviewed literature (e.g. sCBD, 2014; IPCC, 2014a; IPCC, 2014b). The advantage of using a common set of scenarios and models is that they provide a clear and homogenous analysis that may be easier for non-specialists to understand; the disadvantages are that these typically are useful for a very limited range of spatial and temporal scales and decision contexts. The advantages of analyses based on a broad spectrum of published work are that they provide much greater insight into assumptions underlying scenarios and models and their associated uncertainties, and that they address a wide variety of scales and decision contexts because they cover a much larger evidence base. However, very diverse assumptions and indicators used in published work make synthesis difficult (Pereira et al., 2010).

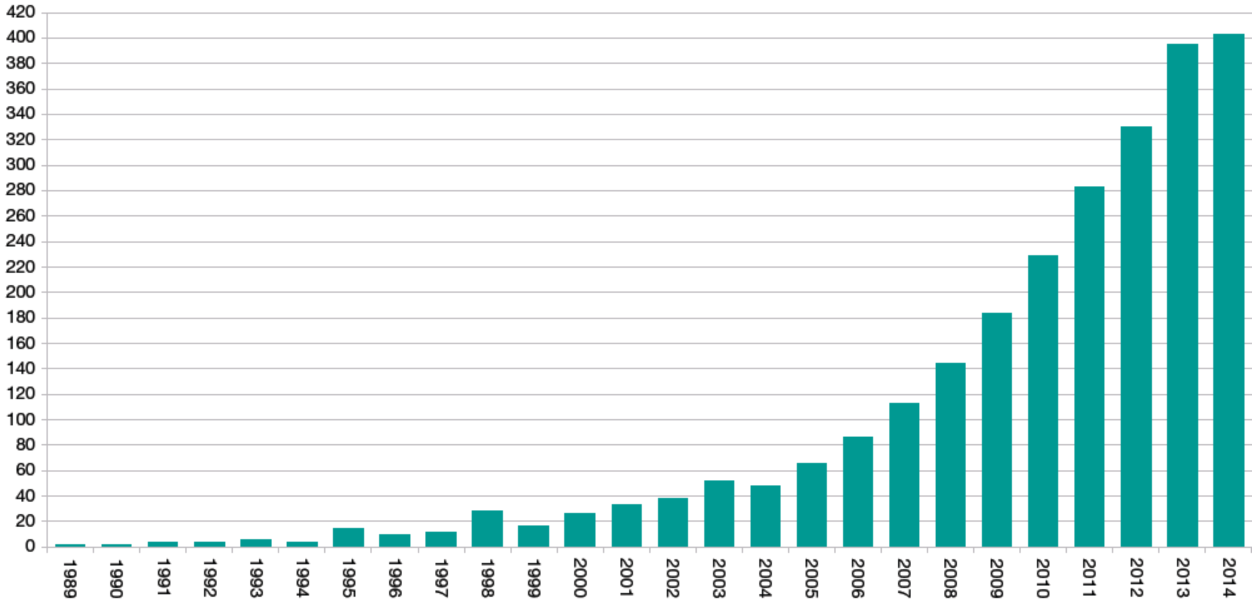



**Figure 1.1:** Change over time in the number of articles published in scientific journals related to future projections of biodiversity and ecosystem services based on scenarios and models. The search pattern used for this analysis has high specificity (a correction for errors of commission has been applied), but is also subject to errors of omission (which are much more difficult to estimate and have not been corrected for). As such, the true number of articles is likely to be substantially higher than indicated here. (Search pattern used 1 Nov 2015 in Web of Science: TS = (Future AND (projection* OR prediction* OR forecast* OR scenario*) AND ('ecosystem service' OR 'ecological service' OR biodiversity OR 'biological diversity' OR 'species richness' OR 'species diversity' OR 'species distribution' OR 'species conservation' OR 'species range' OR 'biological conservation' OR 'nature conservation')). Errors of commission were estimated to be ca. 14% based on a subsample of abstracts, and were substantially higher with older publications. Results were relatively insensitive to the removal of individual search terms with the exception of 'future': removal of this search term led to very high errors of commission).(Modified from FRB 2013)

Despite the use of scenarios and models of biodiversity and ecosystem services in several major global and sub-global assessments, it is difficult to evaluate their role in influencing decision making and popularising biodiversity and ecosystem services, although there is evidence of uptake in national and international policy (Wilson et al., 2014).

More broadly, a variety of factors hamper the more widespread use of scenarios and models in policymaking and management.

These factors include the relatively recent development of scenarios and models for biodiversity and ecosystem services (Figure 1.1); generally insufficient validation of models; insufficient dialogue between scientists and decision makers; and biases in the types of drivers, types of ecosystems, taxonomic coverage of biodiversity, spatial scales and temporal scales (see Section 1.6 and Chapters 2 and 8 for details).

### 1.1.3 Structure of remainder of this chapter

Section 1.2 introduces the fundamental role that models can play in describing relationships between elements of the IPBES Conceptual Framework (1.2.1). It then outlines major types of models of relevance to IPBES activities (1.2.2) and acknowledges the dual contribution that many of the models considered in this assessment (focusing on future change) can also make to assessing past-to-present status and trends in biodiversity and ecosystem services (1.2.3).

Section 1.3 explains how coupling models with scenarios enables the translation of plausible futures for drivers of change and/or alternative policy interventions into expected consequences for biodiversity and ecosystem services (1.3.1). It then outlines major types of scenarios and their relationship with different phases of the policy cycle (1.3.2).

Section 1.4 describes how scenarios and models can inform policy and decision making through a variety of assessment or decision-support interfaces.

Section 1.5 explains the importance of matching employed scenarios, models and interfaces to the needs of different policy or decision-making contexts (1.5.1). It then presents examples of the effective use of scenarios and models of biodiversity and ecosystems services in previous assessment and decision-support activities (1.5.2).

Section 1.6 highlights the need to better recognise, understand and address the current limitations of scenarios and models of biodiversity and ecosystem services, including deficiencies in the spatial, environmental and thematic coverage of existing scenarios and models (1.6.1), gaps in the availability of underpinning knowledge and data (1.6.2), and challenges in dealing with uncertainty (1.6.3).

Section 1.7 outlines the chapter structure of the remainder of the report.

## 1.2 Describing relationships between elements of the IPBES Conceptual Framework with models

### 1.2.1 Overview

The IPBES Conceptual Framework (Figure 1.2; Díaz et al. (2015) provides a logical starting point for introducing and explaining the respective roles of scenarios and models within the context of IPBES.

This framework emerged from an extensive process of consultation and negotiation, leading to formal adoption by the second IPBES Plenary (IPBES/2/4), and therefore represents a key foundation for all IPBES activities. It is a simplified representation of the complex interactions between the natural world and human societies. IPBES recognises and considers different knowledge systems, including indigenous and local knowledge systems, which can be complementary to those based on science. The Conceptual Framework is therefore intended to serve as a tool for achieving a shared working understanding across the different disciplines, knowledge systems and stakeholders that are expected to be active participants in IPBES.

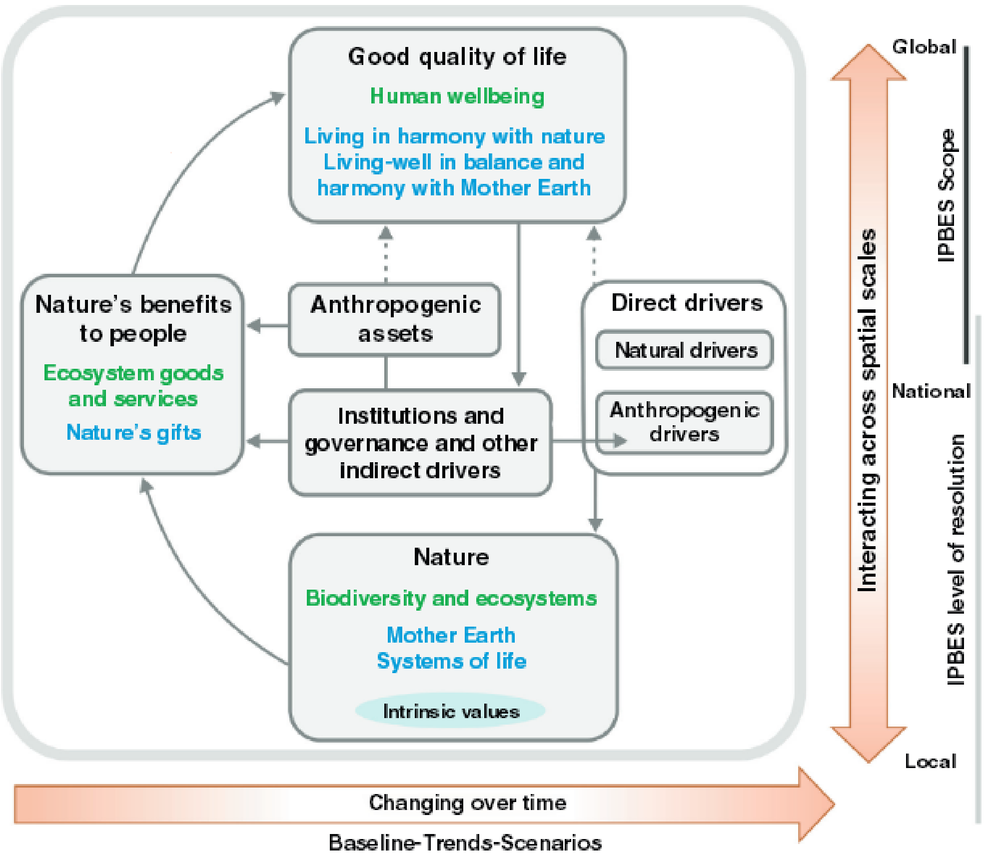



**Figure 1.2:** The IPBES Conceptual Framework, modified from Díaz et al. (2015). This depicts the main elements and relationships for the conservation and sustainable use of biodiversity and ecosystem services, human well-being and sustainable development. Similar conceptualisations in other knowledge systems include 'living in harmony with nature' and 'Mother Earth', among others. In the main panel (delimited in grey), 'nature', 'nature's benefits to people' and 'good quality of life' (indicated as black headlines) are inclusive of all these world views; text in green denotes the concepts of science; and text in blue denotes those of other knowledge systems. Solid arrows in the main panel denote influence between elements; the dotted arrows denote links that are acknowledged as important but are not the main focus of the Platform. The thick coloured arrows below and to the right of the central panel indicate different scales of time and space.

As explained by Díaz et al. (2015), this framework provides a conceptual foundation for the science-policy interface through which knowledge from science and other knowledge systems flows through to policy and decision making via the four main functions of IPBES: knowledge generation, assessment, policy support and capacity building.

Models can make a significant contribution to enabling the flow of data and knowledge to policy and decision making by explicitly describing interactions between major elements of the IPBES Conceptual Framework (Figure 1.3).

In the original framework (Figure 1.2), arrows are used simply to indicate the existence of relationships between elements but convey very little about the precise nature of these relationships. The arrows linking elements in this framework therefore collectively constitute a conceptual model. Replacing these conceptual links with more quantitative descriptions of each of these relationships allows observed or projected changes in the state of one element to be used to estimate or project resulting changes in other elements.

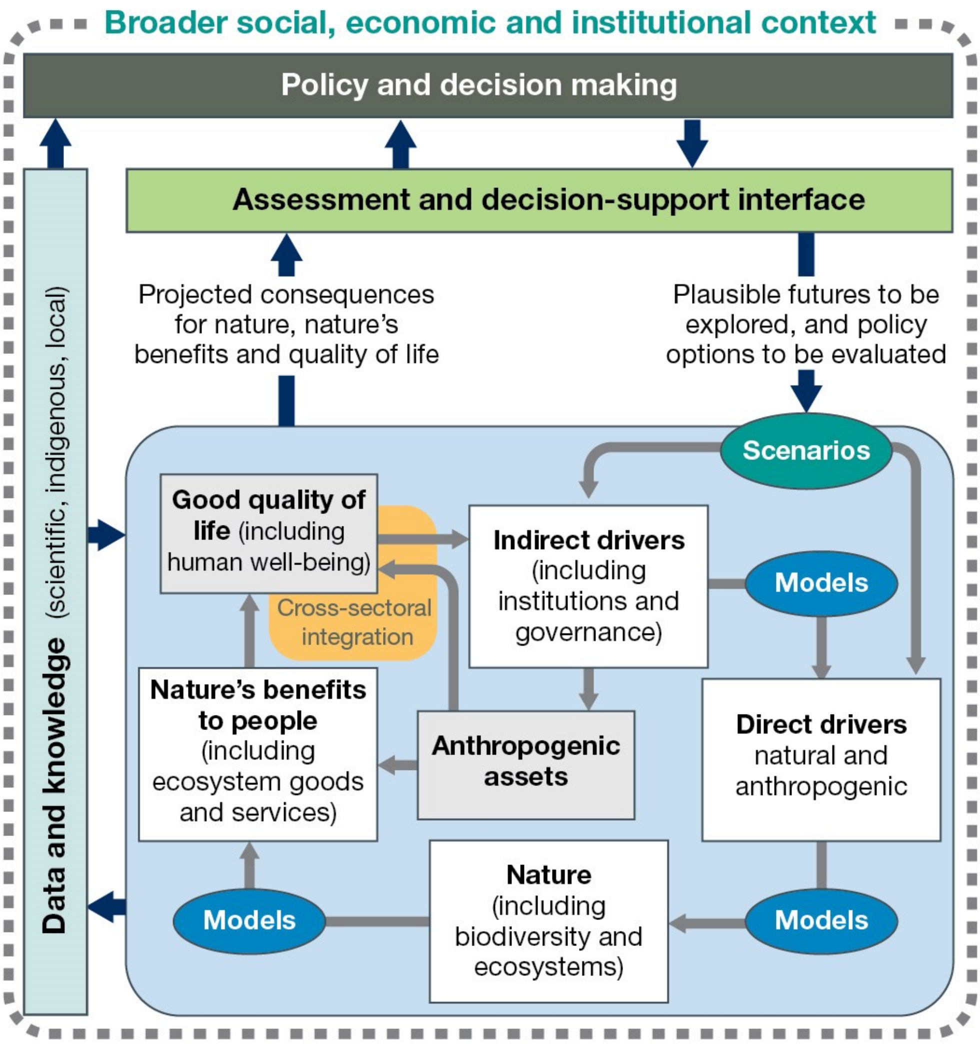



**Figure 1.3:** High-level roles of scenarios and models in assessment and decision support. The rectangular boxes in the lower blue-shaded portion of the diagram represent key elements from the IPBES Conceptual Framework (see Figure 1.2; but note that in the current figure, due to space constraints, elements are translated only into terms commonly used in the scientific literature, e.g., 'Nature' into 'biodiversity & ecosystems', and terms used in other knowledge systems are not depicted). The models addressed in this report focus mostly on relationships between the white-shaded elements. Scenarios and models are directly dependent on data and knowledge for their construction and testing, and add value by synthesizing and organizing this knowledge (box and arrows on left). They usually contribute to policy and decision making through some form of 'interface' – i.e., assessments, formal decision-support tools, or informal interactions (boxes and arrows at top). This interface manages the translation of high-level policy and decision-making needs into explicit scenarios describing plausible futures for drivers of change and/or alternative policy interventions. Models are then used to evaluate these scenarios in terms of expected consequences for nature and nature's benefits to people. The 'cross-sectoral integration' element added to this framework signifies that any comprehensive assessment of human well-being and good quality of life is likely to require integration of modelling across multiple sectors (e.g., energy, health), thereby dealing with a broader set of relevant goals and values than those mediated exclusively by biodiversity or ecosystems. The elements and relationships depicted in this figure are essentially the same as those depicted in Figure SPM.1 in the Summary for Policymakers, even though the latter splits this content across two panels (Modified from Díaz et al., 2015).

## 1.2.2 Types of models of relevance to IPBES activities

A diverse range of models are of potential relevance within the context of IPBES. These models vary in two main ways:

- *What relationships are modelled* – i.e. the outputs or 'response variables' of interest and the inputs used to predict or project these outputs;
- *How these relationships are modelled* – i.e. the way in which the link between input and output variables is represented.

### 1.2.2.1 What relationships are modelled

The models considered in this methodological assessment address three main types of relationship within the IPBES Conceptual Framework (Figures 1.3 and 1.4):

- **Models addressing the effects of changes in indirect drivers** (e.g. socio-political, economic, technological and cultural factors) **on direct drivers** of change in nature (e.g. land-use change, fishing pressure, climate change, invasive alien species, nitrogen deposition);
- **Models addressing the impacts of changes in direct drivers on nature**, including biodiversity and ecosystem functioning; and
- **Models addressing the consequences of changes in nature for the benefits that people derive from nature**, and that therefore contribute to good quality of life (human well-being) – including, but not limited to, ecosystem goods and services.

Models addressing the effects of changes in indirect drivers on direct drivers are often developed for a wide range of purposes that are not expressly intended for use in modelling impacts on nature or nature's benefits to people. Where modelling of direct drivers has already been undertaken by communities of practice within other domains, such as climate modelling or land-use modelling, then resulting projections of these drivers can serve directly as inputs to biodiversity and ecosystem models. In this situation, existing projections of direct drivers function effectively as scenarios of possible futures for the purposes of modelling consequences for nature and nature's benefits (Figure 1.3). In other situations, interest may be focused on modelling consequences of scenarios of indirect drivers, rather than direct drivers. In this case, modelling of the effects of indirect-driver scenarios on direct drivers will need to be undertaken as a first step in modelling consequences for nature or nature's benefits. Such models are therefore covered in Chapter 3 of this assessment. It should be noted that the development of scenarios of indirect drivers also often involves models of various types, including human demographic models, governance models, economic models and agent-based models describing the behaviour of social systems (Figure 1.4). However, for the purpose of this assessment, it is assumed that such modelling will typically be undertaken outside the core domain of IPBES (see Chapter 3 for further explanation).

A tremendous variety of variables are simulated, and thereby predicted or projected, by models of nature (biodiversity and ecosystem functioning) and nature's benefits (Figure 1.4).

In some cases, only a single variable is simulated. For example, species distribution models are often used to predict the spatially-explicit response of just one variable (such as species presence or absence) to environmental change. In other cases, models predict multiple variables, but typically only a small subset of the variables listed in Figure 1.4. For example, biodiversity models simulate dynamics of genes, species, functional groups or communities, but most focus on only one of these levels and none simulate biodiversity dynamics at and between all these levels.

In practice, relationships between variables linking the three main components of nature and nature's benefits differ greatly in the frequency and detail with which they are treated in the scientific literature and assessments (Figure 1.4). For example, models of ecosystem function, especially at large spatial scales, typically represent biodiversity using a small number of groups of species that have similar characteristics (i.e. functional groups). A few models of ecosystem function use species-level variables, but very few incorporate variables related to genetic adaptation (but see Kramer et al., 2010). Models of nature's benefits typically rely on empirical relationships between habitat type and ecosystem services (arrow directly from habitat) or use inputs from variables simulated by models of ecosystem function, but few account for the contribution of species diversity to ecosystem function (Cardinale et al., 2012), but note that some models do account for a small set of key species interactions).

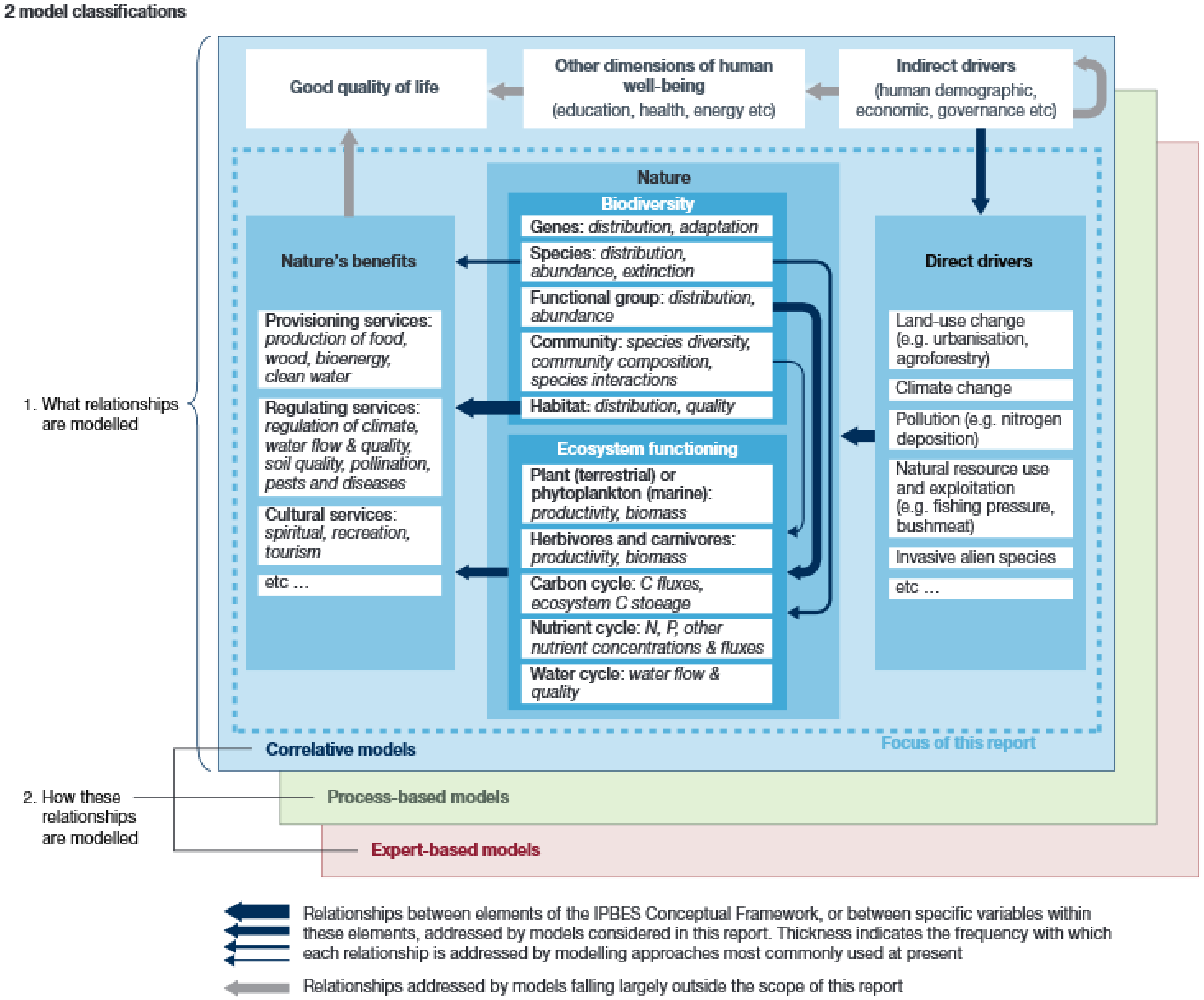



**Figure 1.4:** Major types of models of relevance to IPBES activities, classified according to 'what relationships are modelled' (represented by the arrows linking elements of the IPBES Conceptual Framework, or variables within these elements) and 'how these relationships are modelled' (represented by the light-blue, green and pink-shaded panels). All of the relationships depicted on the light-blue-shaded 'correlative models' panel can also be modelled using 'process-based models' (green-shaded panel) or 'expert-based models' (pink-shaded panel).

Modelling of nature's benefits to people can serve as a key input to assessing human well-being, and therefore good quality of life (Figures 1.3 and 1.4). Such assessments will, however, typically require broader consideration, and therefore modelling, of dimensions of human well-being beyond those mediated primarily by biodiversity or ecosystems, such as education, health and energy.

Modelling of these other dimensions is most often undertaken within domains or sectors largely external to that of IPBES, and these models are therefore not covered in any detail by this report. However, the report does recognise the growing need for the cross-sectoral integration of models, trade-offs and synergies between these dimensions (e.g. Hilderink and Lucas, 2008), particularly within the context of the United Nations' recently ratified Sustainable Development Goals (SDGs) (https://sustainabledevelopment.un.org/; see also Section 1.4, Chapters 2, 5 and 6).

As depicted in Figures 1.3 and 1.4, models of drivers, nature and nature's benefits can be implemented as a linked chain, where the input for one model is derived from the output of the previous model in the chain (e.g. Bateman et al., 2013; Nelson et al., 2009). Increasingly, however, models such as these are being integrated even more strongly by treating them as components of a single modelling framework, thereby enabling the more effective consideration of interactions and feedbacks between these components. Examples of this level of integration include end-to-end ecosystem models (e.g. Fulton, 2010) and integrated assessment models (e.g. Stehfest et al., 2014).

Integrated assessment models (IAMs) combining modelling of multiple environmental, social and economic system components are increasingly being used in global and regional assessment activities.

Integrating very different types of knowledge within IAMs is particularly challenging (e.g. De Vos et al., 2013), but necessary if these approaches are to provide an effective foundation for assessing human well-being and quality of life. While IAMs usually account for at least some ecosystem functions and services, they often exclude key ecosystem functions and omit cultural services, and generally lack representation of biodiversity below the functional group or habitat type level (Harfoot et al., 2014a), but see Alkemade et al. (2009) for examples of including species diversity in global integrated models). Regardless of the precise approach used to link or integrate models, great care needs to be taken to account for propagation of error, consistency of variables, differences in spatial and temporal resolution, and costs and benefits of increasing complexity (see Chapter 6).

#### 1.2.2.2 How relationships are modelled

The relationship between input and output variables can be represented or described by a model in many different ways, both quantitative and qualitative (Börner et al., 2012; Ritchey, 2012).

Three broad approaches to modelling relationships between input and output variables are recognised throughout this assessment (Figure 1.4, and see Chapter 4 for further explanation):

- **Correlative models**, in which available empirical data are used to estimate values for parameters that do not have a predefined ecological meaning, and for which processes are implicit rather than explicit;
- **Process-based models**, in which relationships are described in terms of explicitly-stated processes or mechanisms based on established scientific understanding and model parameters therefore have a clear, predefined, ecological interpretation;
- **Expert-based models**, in which the experience of experts and stakeholders, including local and indigenous knowledge holders, is used to describe relationships.

Correlative modelling is probably the best known, and most widely applied, of these three approaches, due largely to the popularity of correlative species distribution modelling in recent years (Elith and Leathwick, 2009).

Process-based modelling encompasses a wide range of techniques, many of which represent underlying processes using mathematical equations, for example the modelling of population and meta-population dynamics (e.g. Brook et al., 2000; Gordon et al., 2012) and of ecosystem function (e.g. Harfoot et al., 2014b). Other techniques in this class represent underlying processes as quantitative rules rather than as equations, for example rule-based modelling to inform extinction risk assessment (e.g. Mace et al., 2008).

Expert-based modelling also encompasses a wide variety of techniques, in this case for capturing and representing expert knowledge of relationships between variables of interest (e.g. Priess and Hauck, 2014; Walz et al., 2007). In this context, an 'expert' is considered to be anyone who has acquired good knowledge of a subject through his or her life experience (Kuhnert et al., 2010), including local or indigenous knowledge holders in addition to scientists. It is assumed, however, that the expert is a reliable source of information within a specific domain (Burgman, 2005).

Some modelling techniques allow these different approaches to be combined within a single model. For example, in Bayesian Belief Networks, expert-based knowledge can be combined with information derived through correlative or process-based approaches (Haines-Young, 2011).

A variety of modelling approaches may often be available for addressing a particular question. The position taken throughout this methodological assessment is that there is usually no single best modelling approach for any given application. In particular, debates about the use of models working with correlative versus process-based versus expert-based models are frequently polluted by misconceptions about the usefulness of these various types of models. Many modelling exercises have clearly illustrated the benefits of examining multiple model types in terms of understanding of underlying processes, improving the ability to simulate biodiversity and ecosystem functions, providing complementary sets of variables and estimating uncertainty (Cheaib et al., 2012; Gritti et al., 2013; van Oijen et al., 2013). The use of multiple models does not necessarily require quantitative comparisons among models. However, in some cases IPBES may want to stimulate work on quantitative multi-model comparisons since, as the IPCC has amply demonstrated for climate models and some models of impacts on ecosystems (IPCC, 2014a), these can often carry more weight in decision making than individual models. This does not mean that all models are equally good. As such, models need to be thoroughly tested with independent data and an evaluation of the strengths and weaknesses of models should ideally be included when presenting model outcomes. The following chapters provide more specific guidelines for selecting models and for evaluating their strengths and weaknesses.

### 1.2.3 Using models to assess past-to-present status and trends

This assessment focuses primarily on the use of models, in conjunction with scenarios, to explore potential changes in nature and nature's benefits into the future. However, before adopting this particular focus throughout the remainder of this report, it is worth noting that modelling can, and does, also play an important role in assessing status and trends even in the absence of scenarios.

All of the approaches outlined above require, as input, information on the state of one element of the IPBES Conceptual Framework, which a model then uses to predict, or project, the state of another element. These models can therefore be applied either to future projections of input variables (based on scenarios; see Section 1.3) or to actual observations (data) for these same input variables (Figure 1.5). The latter option can help to shed valuable light on the present status of nature and its benefits, and on changes or trends in this status past-to-present (Leadley et al., 2014). Several elements of the IPBES

Conceptual Framework align well with major categories of indicators within the widely adopted 'drivers-pressures-states-impacts/benefits-responses' (DPSIR) approach to status-and-trend assessment (Feld et al., 2010; Sparks et al., 2011). Modelling can add considerable value to such assessments in two important ways:

- Modelling can help to fill gaps in the data needed to underpin key indicators. While ongoing data acquisition is clearly of vital importance (see Section 1.6 and Chapter 8), data are much easier and/or less costly to obtain for some elements of the IPBES Conceptual Framework than for others. For example, advances in remote sensing have now made it possible to track temporal changes in a number of direct drivers (pressures), including habitat conversion and climate change, at relatively fine spatial resolutions across extensive regions (Hansen et al., 2013). On the other hand, most components of biodiversity, particularly at the species and genetic levels, are not detectable through remote sensing, and changes in their state can be observed only through direct field survey. Such data therefore tend to be sparsely and unevenly distributed across both space and time. Modelling offers a cost-effective means of filling gaps in this coverage by using remotely derived, and therefore geographically complete, information on drivers to estimate changes in the state of biodiversity (past to present) expected across unsurveyed areas (Ferrier, 2011; Leutner et al., 2012; Turner, 2014). Using modelling to fill gaps in information can play an equally valuable role in assessing status and trends in nature's benefits to people, for example by estimating changes in the supply of ecosystem services, relative to the distribution of people receiving these benefits, from remotely-sensed land cover classes and structural or functional ecosystem attributes (biomass, net primary production, etc.) (Tallis et al., 2012; Andrew et al., 2014).
- Modelling can provide a process-based alternative to the use of composite indicators in integrating multiple pressure-state-response indicators. Applications of the DPSIR framework typically generate large numbers of indicators (Butchart et al., 2010; Sparks et al., 2011), distinguished not only by their focus on different high-level components of this framework (e.g. pressure indicators versus state indicators versus response indicators) but also by differences in the focus of indicators within each component (e.g. indicators of habitat conversion pressures versus invasive alien species pressures, or indicators of habitat protection (reservation) responses versus invasive species control responses). To provide a better sense of the overall status of, and trends in, the condition or 'health' of the system as a whole, these individual indicators are sometimes aggregated to produce one, or a small number of, composite indicators or indices (e.g. Halpern et al., 2012). While aggregation will often be most readily achieved through simple summation or multiplication (Butchart et al., 2010), this may fail to adequately address the often complex, non-linear nature of interactions between multiple pressure, state and response elements in real-world systems. Modelling offers an alternative means of integrating data and indicators, describing past-to-present changes across multiple system elements, and thereby better accounting for complexities and dynamics in these interactions (Vackar et al., 2012; Pereira et al., 2013; Tett et al., 2013).

# 1.3 Coupling models with scenarios to explore future possibilities and options

## 1.3.1 Overview

Policy and decision-making processes often require looking beyond the present to the future. Questions raised in these processes might include: What is the risk of future loss of nature, or nature's benefits to people? How would alternative policy or management interventions alter this outcome? Using models to address questions relating to possible changes in the future, rather than to actual changes in the present or recent past, poses special challenges. In this situation, observations of change (e.g. in drivers) are not available to use as inputs to models because these changes are yet to occur. Furthermore, there is often considerable uncertainty associated with the future trajectory of any given input variable because this trajectory will be affected by events and decisions that have also not yet occurred, and are often highly unpredictable. Scenarios provide a useful means of dealing with the reality that not just one, but many, futures are possible (Pereira et al., 2010; Cook et al., 2014).

Scenarios and models play different, but highly complementary, roles in informing and supporting policy and decision making (Figures 1.3 and 1.5). Scenarios are used to describe plausible futures for drivers of change, and options for altering the course of these drivers through policy and management interventions. Models then enable scenarios of change in drivers to be translated into expected consequences for nature and nature's benefits to people.

### 1.3.2 Types of scenarios relating to different phases of the policy cycle

What exactly is meant by 'policy and decision making'? The adoption of this term in Figure 1.3 follows its use in various other IPBES documents including, for example, documentation of the Conceptual Framework (Decision IPBES-2/4, http://www.ipbes.net/). However, policy and decision making can encompass a very broad range of processes and activities conducted in a wide variety of contexts across multiple scales.

Numerous frameworks have been proposed over recent decades for conceptualising phases or elements of the policy cycle, and similar frameworks have also been developed for describing adaptive planning or management cycles. There is considerable commonality between most of these frameworks. For the purposes of this assessment, four broad phases of the policy cycle are recognised (see Chapters 2 and 3 for further detail): 1) agenda setting, 2) policy design, 3) policy implementation (also referred to as 'planning and management' in parts of the report), and 4) policy review.

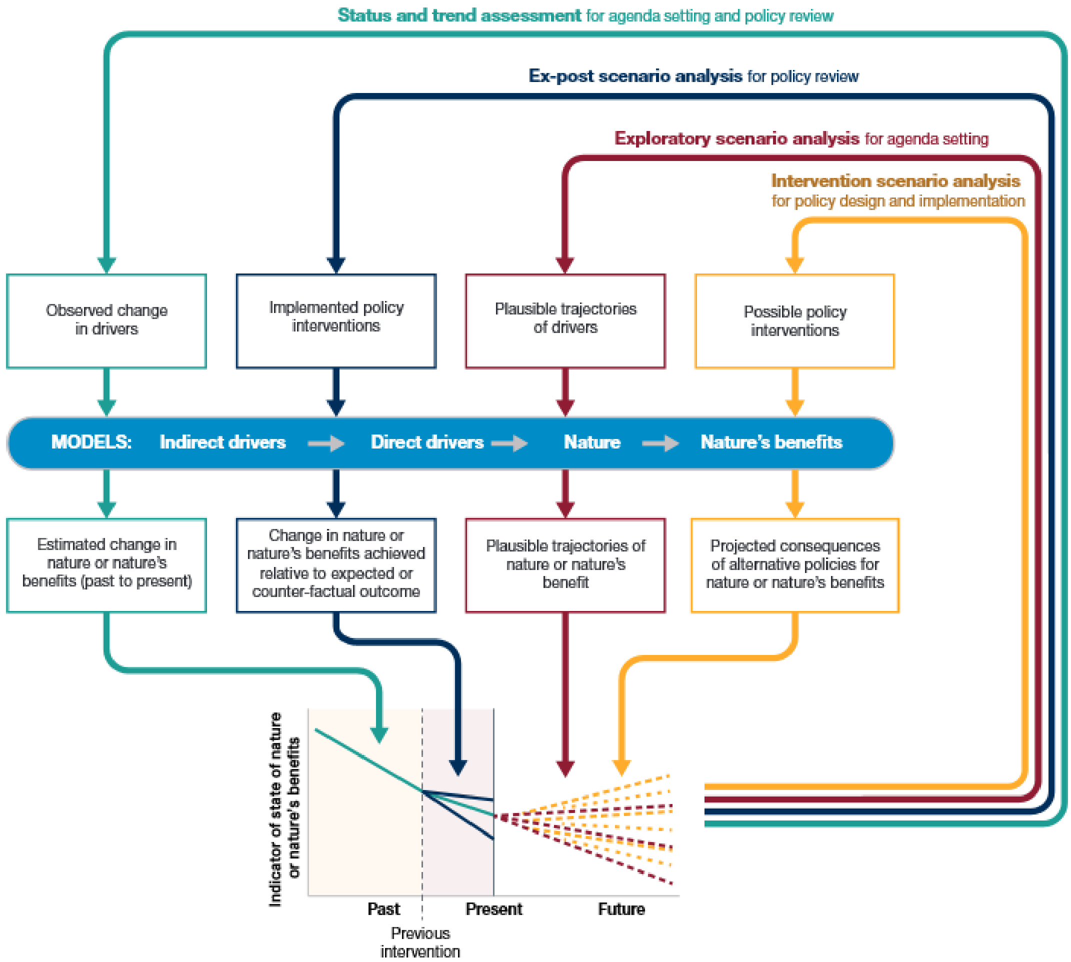



**Figure 1.5:** Major ways in which models and scenarios can be combined to inform agenda setting, policy design, policy implementation, and policy review. Models estimate or project changes in nature and nature's benefits as a function of: observed changes in drivers, for status-and-trend assessment (depicted in blue-green); plausible trajectories of drivers, for exploratory scenario analysis (depicted in red); possible policy interventions, for intervention scenario analysis (depicted in orange); or implemented policy interventions, for ex-post scenario analysis (depicted in purple).

Scenario analysis and modelling can inform and support activities across all four of these phases. As depicted in Figure 1.5, this involves using different types of scenarios of drivers and policy interventions as inputs to a common set of models for assessing the expected consequences of these scenarios for nature and nature's benefits. Various terminologies and typologies for describing and classifying these different types of scenarios have been proposed and used in the scenario literature (see for example van Notten et al. (2003) and van Vuuren et al. (2012)).

This assessment deals primarily with two broad types of scenarios, referred to throughout this report as:

- **Exploratory scenarios** (also known in the literature as 'explorative scenarios' or 'descriptive scenarios') that examine a range of plausible futures based on potential trajectories of drivers – either indirect (e.g. socio-political, economic and technological factors) or direct (e.g. habitat conversion, climate change);
- **Intervention scenarios** (also known in the literature as 'policy scenarios') that evaluate alternative policy or management options – either through **target seeking** (also known as 'goal seeking' or 'normative scenario analysis') or through **policy screening** (also known as 'ex-ante assessment').

Scenarios of a third broad type depicted in Figure 1.5 receive less attention in this assessment. These are policy-evaluation scenarios employed in ex-post assessments of the extent to which outcomes actually achieved by an implemented policy match those expected based on modelled projections, thereby informing policy review (see Chapter 3 for some further discussion of this scenario type).

#### 1.3.2.1 Exploratory scenarios

Exploratory scenarios are employed mostly in the agenda-setting phase of the policy cycle. A sizeable proportion of previous efforts in the scenario analysis and modelling of biodiversity and ecosystem services have used exploratory scenarios to identify and promote the need for action and opportunities to address detrimental changes in nature and its benefits.

This use of exploratory scenarios in agenda setting can add considerable value to the assessment of status and trends described in Section 1.2.3, by extending the focus of assessment from changes that are known to have already occurred past-to-present, to changes that might occur into the future (Pereira et al., 2010; Cook et al., 2014). At its most basic, this extension may simply involve the statistical extrapolation of observed trends in the state of biodiversity and ecosystem services into the future, assuming that levels or rates of change in underlying drivers will remain constant (e.g. Tittensor et al., 2014).

To more explicitly consider uncertainties in the future trajectories of drivers, exploratory scenarios are most commonly formulated as a discrete set of ‘plausible futures’, specified as narratives or storylines of economic and socio-political pathways, and including assumptions regarding, for example, technological development (Spangenberg et al., 2012). The formulation of plausible futures may involve the use of techniques such as horizon scanning to help identify future problems, threats and opportunities at the margins of current thinking and planning (Cook et al., 2014). Examples of this general approach are the IPCC’s Special Reports on Emission Scenarios and similar sets of scenarios employed in the Millennium Ecosystem Assessment and the Global Environment Outlooks. In recent years, the plausible futures approach has been increasingly complemented by alternative approaches to the development of exploratory scenarios. For example, ‘probabilistic scenarios’ can be developed using similar process-based models to those employed in modelling plausible futures, but using inputs drawn from probability distributions for each parameter based on best-available empirical data or expert knowledge, in place of discrete ‘plausible’ combinations of parameter values, thereby allowing probabilities to be attached to resulting projections (e.g. Abt Associates, 2012).

#### 1.3.2.2 Intervention scenarios

Moving from assessing the need for action in agenda setting to actual decision making around specific actions in policy design and implementation shifts the focus of scenario analysis and modelling from exploratory scenarios to intervention scenarios.

While the boundary between policy design and implementation is often rather fuzzy, the requirements for intervention scenarios at either end of this spectrum can be quite different, especially in terms of the level of specificity and spatial explicitness with which potential actions are defined. This is particularly the case for policies allowing choice in the location of actions implemented under these policies – for example the establishment of new protected areas to meet a high-level target (e.g. 17% of terrestrial area, as specified by Aichi biodiversity target 11), or the allocation of funding under various economic instruments (e.g. an environmental stewardship scheme). In such situations, lower-level decisions made during the implementation of a high-level policy can have significant implications for the effectiveness of the outcome actually achieved by that policy – not just in biophysical terms, but also in terms of implementation costs and socio-economic consequences for people affected by these decisions. For example, decision making around the precise location of new protected areas or funded stewardship actions may require spatially-explicit intervention scenarios at a much finer spatial resolution than those needed to inform the initial design of these high-level policies.

Two quite different strategies can be used to develop and evaluate intervention scenarios – target seeking, and policy screening (van Vuuren et al., 2012). Target seeking, in which a desired endpoint or goal is first defined and analytical techniques such as backcasting (Dreborg, 1996) are then used to search for intervention scenarios that fulfil this goal, is increasingly being employed to inform high-level policy design (see Box 1.1 in Section 1.5.2 for an example of the application of this strategy, for the Rio+20 conference, by PBL (2012)). Policy screening, in which options for policy or management intervention are defined in advance and the relative effectiveness of these options is then evaluated through forecasting, is employed widely for both policy design and implementation (see Box 1.2 in Section 1.5.2 for an example of this strategy). These two strategies are discussed further in Section 1.4 and in Chapters 2 and 3.

The distinction between exploratory scenarios and intervention scenarios is often not as clear-cut as the above descriptions might suggest. Scenario analyses of biodiversity and ecosystem services are increasingly integrating elements of both exploratory and intervention scenarios within a single analysis.

The exploratory component of such analyses provides a means of addressing uncertainties associated with drivers that might affect the outcome of a given policy or decision-making process but are external to, and therefore not amenable to control or influence by, that process (Peterson et al., 2003). These drivers are therefore viewed as being 'exogenous' to the particular policy or decision context (Chermack, 2011). The intervention component then focuses on drivers that can be influenced by this particular process and are therefore regarded as 'endogenous' or 'policy-relevant' (*ibid.*). Exogenous drivers typically operate over broader spatial and temporal extents than those targeted by policy interventions addressing endogeneous drivers. For example, in developing a national policy to protect or restore habitat to enhance the persistence of biodiversity under climate change, modelling of outcomes for biodiversity might be undertaken for integrated scenarios that pair alternative protection or restoration options at the national scale (the intervention component) with plausible climate futures at global or regional scales (the exploratory component). This approach would thereby account for uncertainties associated with exogenous drivers of climate change when assessing policy options addressing endogenous drivers of habitat degradation. Considerable potential now exists to further combine integration of different types of scenarios across multiple spatial and temporal scales with integration of models dealing with multiple elements of the IPBES Conceptual Framework (e.g. through IAMs; section 1.2.2.1) as depicted in Figure 1.6.

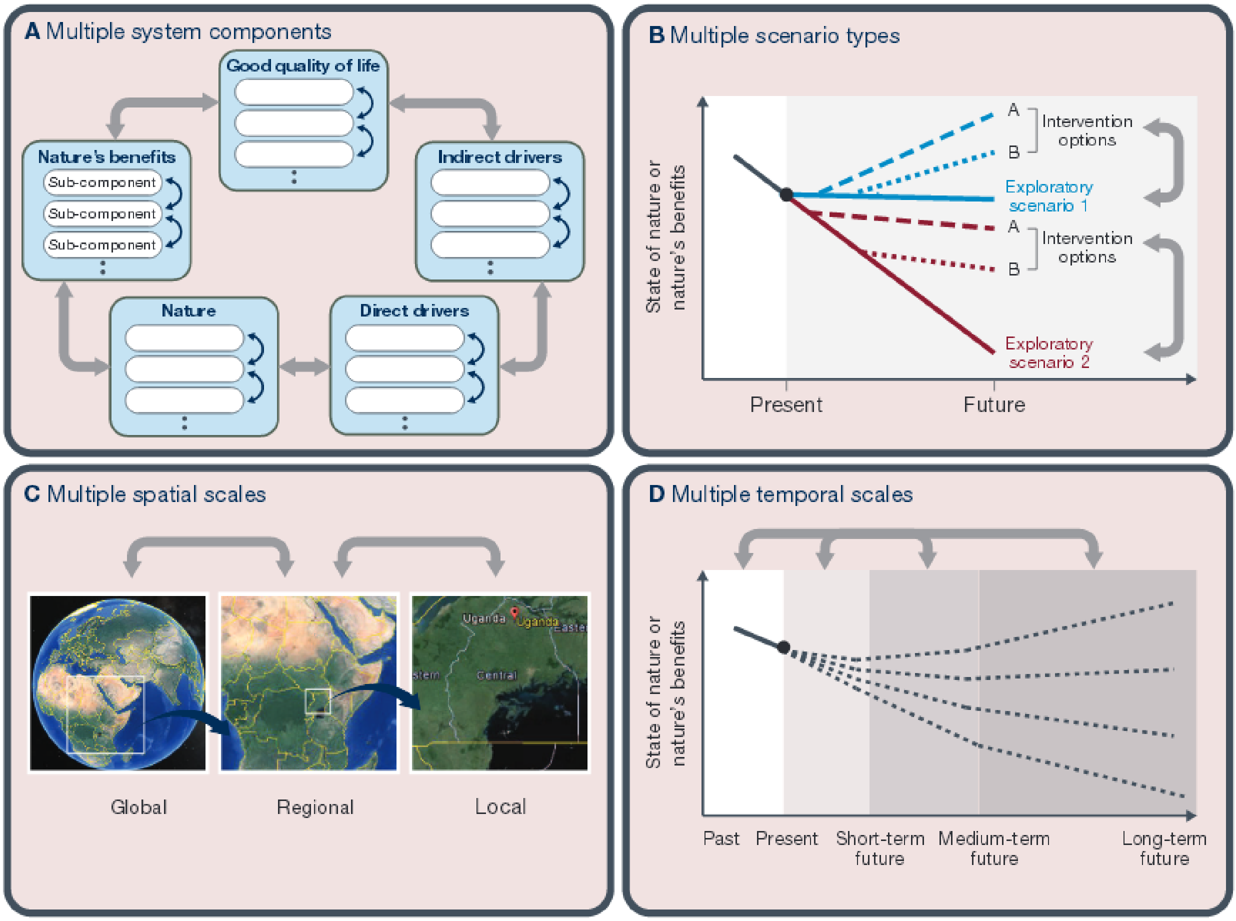



**Figure 1.6:** Linking scenarios and models in four key dimensions: system components, scenario types, spatial scales and temporal scales, with the thick grey arrows indicating linkages within each dimension. Panel A illustrates linkages between scenarios and models across the different components of the IPBES Conceptual Framework (thick grey arrows) as well as between their sub-components (thin blue arrows; for example linking biodiversity with ecosystem function sub-components of nature). Panel B shows ways in which different types of scenarios, such as exploratory and intervention scenarios, can be linked. Panel C indicates linkages across spatial scales from local to global. Panel D illustrates linking the past, present, and several time horizons in the future (dashed lines indicate a range of exploratory scenarios). Two or more of these dimensions of linkages can be used in combination (e.g., linking different types of scenarios across spatial scales).

## 1.4 Linking scenarios and models to policy and decision making through assessment and decision-support interfaces

### 1.4.1 Overview

The interaction of policy and decision-making processes with scenarios and models will usually be mediated by some form of assessment or decision-support system or process, here referred to generically as an 'interface' (Figure 1.3). It is through this interface that high-level policy and decision-making needs are translated into explicit scenarios for analysis by appropriate models and, in turn, that outputs from this modelling are interpreted and communicated back to the world of policy and decision making.

The form and complexity of the interface needed for any given application depends very much on the precise nature of the policy or decision-making process being served, and particularly on the phase of the policy cycle being addressed (from Section 1.3.2 above). For processes focused on agenda setting, this interface may simply involve selecting and formulating any exploratory scenarios to be assessed, managing the analysis of these scenarios using an appropriate set of models, and reporting results from these analyses in terms of projected outcomes for nature or nature's benefits to people. The interface employed in such situations will therefore often take the form of an 'assessment', typically communicating results in technical reports and/or published papers.

### 1.4.2 Decision-support interfaces

Managing the application of intervention scenarios to policy design and implementation, as opposed to agenda setting, often requires a shift from relatively static assessment to more dynamic, and interactive, decision support (see Chapter 2).

This is because the number of potential options for intervention can be very large, particularly in the policy implementation phase. In terms of the examples from Section 1.3.2.2, for example, this means a large number of possible configurations of protected areas or of funded stewardship actions.

If all possible options of interest are known at the outset of a decision-making process then various forms of mathematical (computer-based) optimisation might be used to automate the search for an intervention or set of interventions that either maximises the expected outcome for nature or nature's benefits, or maximises the robustness of this outcome in the face of future uncertainties (Williams and Johnson, 2013). However, many policy design and implementation processes – especially at lower (more local) levels of decision making – require consideration of intervention options that are not necessarily known in advance but instead arise dynamically from interactions and negotiations within the process itself. This means that intervention scenarios must be formulated, and analysed, progressively throughout the decision-making process. Searching for, and reaching agreement on, effective policy or management interventions in such situations becomes more a process of interactive trial and error, involving adaptive evaluation and the modification of intervention scenarios informed by feedback on the modelled consequences of these options. Growing recognition since the 1970s (Holling, 1978) of this need for the more interactive, and inclusive, involvement of decision makers and stakeholders in the formulation and evaluation of intervention scenarios is reflected in the recent proliferation of planning approaches, both qualitative and quantitative, based around 'participatory scenarios' (Walz et al., 2007; Sandker et al., 2010; Priess and Hauck, 2014).

The basic idea of using models to evaluate consequences of intervention scenarios as a foundation for decision making is already well established within several existing methodological paradigms or frameworks including, for example, 'management strategy evaluation' (De la Mare, 1998; Fulton et al., 2014), 'structured decision making' (Addison et al., 2013), 'scenario planning' (Peterson et al., 2003) and 'strategic foresight' (Cook et al., 2014) (see Chapter 2 for a comprehensive review of such approaches). Tools associated with these, and related, paradigms are often called upon to fulfil the decision-support interface role depicted in Figure 1.3. Linking such tools with scenarios and models offers a highly structured, and potentially very effective, means of implementing the target-seeking and policy-screening strategies introduced in Section 1.3.2.2 for developing and evaluating options for policy or management intervention.

### 1.4.3 Embedding scenarios and models of nature and nature's benefits within

### broader cross-sectoral assessment and decision support

In many policy or decision contexts, the consequences of exploratory and intervention scenarios will need to be evaluated in terms of impacts on multiple values or objectives. These might include different values associated with nature (e.g. multiple biodiversity or ecosystem attributes) or nature's benefits (e.g. multiple ecosystem services). If impacts on such values have been projected using multiple models, the assessment and decision-support interface (depicted in Figure 1.3) may also need to play an important role in aggregating and synthesising modelled outcomes across these values. Various levels of rigour and sophistication can be employed in this integration, ranging from relatively simple visualisation techniques through to more mathematical approaches such as multi-criteria analysis (Arhonditsis et al., 2002).

The breadth of values and objectives to be considered in policy and decision making will often extend well beyond those directly associated with, or mediated by, nature and nature's benefits. This is likely to be the case for many, if not most, assessment and decision-support processes addressing overall human well-being, and therefore quality of life (e.g. Hatfield-Dodds et al., 2015). As already indicated in Section 1.2.2.1, such processes may require results from modelling of nature and nature's benefits to be integrated with modelling of other major dimensions of human well-being (e.g. education, health or energy) undertaken within other domains or sectors (as represented by the 'cross-sectoral integration' element in Figure 1.3). Techniques such as multi-criteria analysis can again play a crucial role in aggregating modelled outcomes across broader sets of values into composite indices of human well-being (e.g. Ding and Nunes, 2014). However, it should be recognised that this level of cross-sectoral integration may often be driven and managed by assessment and decision-support processes external to, or transcending, the domain of IPBES.

In many cases, modelling of consequences of scenarios for nature and nature's benefits, undertaken by communities of practice associated with IPBES, will need to feed into higher-level processes assessing implications for human well-being across a broader range of values and objectives.

Demand for this level of cross-sectoral integration is set to escalate following the recent ratification of the United Nations' (UN) Sustainable Development Goals (https://sustainabledevelopment.un.org/; see also Chapters 2, 5 and 6). The SDGs, now agreed to by Member States of the UN, have ushered in a new set of universal goals and targets ranging from poverty eradication to the sustainable management of natural resources, to be achieved by 2030. Unlike in the previous Millennium Development Goals, both nature and nature's benefits have been recognised as making important contributions to human well-being in the SDGs, and at least 6 of the 17 SDGs are directly linked to aspects of biodiversity and ecosystem services. Scenario analysis and modelling across multiple sectors are likely to play a vital role in monitoring progress in relation to the SDGs, and in ensuring that effective policy instruments and institutional frameworks are put in place to meet the associated targets.

Any use of scenarios and models to inform policy and decision making will typically take place within a much broader – and often highly complex – social, economic and institutional context (Figure 1.3). Policy design and implementation will rarely, if ever, be driven by scenario analysis and modelling alone.

It is therefore important to recognise from the outset of this assessment that guidance provided by scenarios and models will nearly always constitute just one of a number of inputs and considerations shaping policy and management decisions. In addition, the relationships between scenarios, modelling and decision making are often more complex than Figure 1.3 depicts, and can involve highly dynamic interactions and feedbacks between scenario and model development, knowledge and data generation, and engagement with decision makers (see Chapter 8 for a more detailed discussion).

## 1.5 Combining scenarios, models and interfaces in different ways to serve diverse policy and decision-making needs

### 1.5.1 Tailoring approaches for particular policy or decision contexts

It is clear from the scene-setting introductions to models, scenarios and decision-support interfaces provided in Sections 1.2, 1.3 and 1.4 that a considerable diversity of approaches – and of options for applying these approaches – exists across all of these components. How can policy practitioners and scientists seeking to use scenarios and models to inform policy and decision making around nature and nature's benefits choose an appropriate solution from the many alternatives on offer?

An important message emerging from this assessment, and recurring across all chapters of this report, is that the appropriateness of different methodological approaches and options depends very much on the characteristics and needs of any particular policy or decision-making process – in other words on the 'policy or decision context'.

It is therefore vital that approaches employed in different contexts are tailored carefully to the needs of those contexts. No single solution can serve all needs, and different contexts will often require very different solutions.

Figure 1.7 depicts important characteristics and needs of policy and decision-making processes that are likely to vary markedly between contexts. This figure also depicts choices in the selection or design of scenarios, models and decision-support interfaces that depend on these policy context characteristics. While many of these dependencies have already been touched on in previous sections, they are synthesised in Figure 1.7, and further summarised below, to provide readers with a better sense of the overall challenge in ensuring that the employed approaches are well matched to the needs of particular policy or decision contexts.

**Phase of the policy cycle**

Activities aligned with different phases of the policy cycle require the use of different types of scenarios, and different types of assessment or decision-support interfaces. For example, processes focused on agenda setting typically require the use of exploratory scenarios, whereas those focused on policy design or implementation are instead likely to require intervention scenarios (see Section 1.3 and Chapter 3). The interfacing of scenarios and models with agenda setting will often simply take the form of a relatively simple, static assessment in which expected outcomes for nature or nature's benefits are modelled for a discrete set of exploratory scenarios, then documented in a report or publication. On the other hand, the interfacing of scenarios and models with policy design and implementation is more likely to require the use of structured, and often dynamic, decision-support tools to help manage and evaluate large numbers of intervention options (see Section 1.4; Chapter 2).

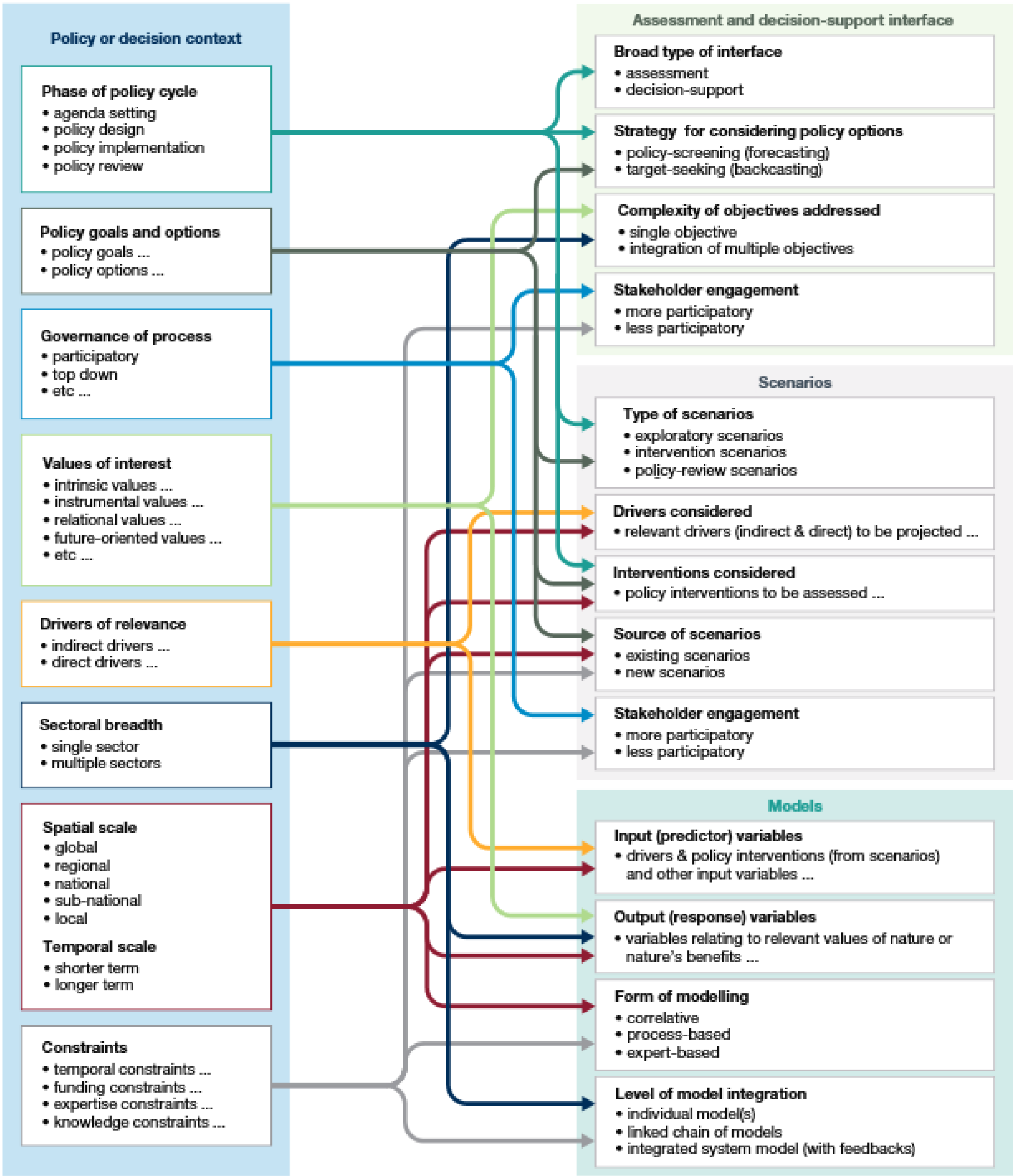



**Figure 1.7:** Dependencies between the characteristics and needs of policy and decision-making processes in different contexts, and the selection or design of scenarios, models, and decision-support interfaces to serve these needs. Each coloured arrow indicates that the selection or design of a particular attribute of 'Assessment & decision-support interface', 'Scenarios', or 'Models' (right side of figure) is dependent on a particular characteristic or need of the 'Policy or decision context' (left side of figure).

## Policy goals and options

The way that goals and options are defined in any given policy design or implementation process has a strong bearing on the appropriateness of target-seeking versus policy-screening strategies for developing and evaluating intervention scenarios (see Sections 1.3 and 1.4; Chapters 2 and 3). Processes focused on identifying possible policy pathways for achieving a clearly defined target or set of targets (e.g. the Convention on Biological Diversity's (CBD) Aichi biodiversity targets, or targets associated with the Sustainable Development Goals) are likely to be best served through the employment of a target-seeking strategy. Other processes may, however, simply involve choosing between a set of predefined policy or management options, and are therefore better served through policy screening.

**Spatial and temporal scale**

Activities across all policy-cycle phases can occur at a wide range of spatial scales – global, regional, national, sub-national and local. The spatial extent (coverage) and resolution (grain or detail) of scenarios and models employed in any policy or decision-making process must therefore be aligned carefully with the scale of interest for that process. Such processes can also address quite different temporal scales of concern – ranging from processes focused on short-term outcomes (changes made over a few years) through to those focused on achieving longer-term change (e.g. over several decades) – which again has strong implications for the temporal scale of any scenarios and models employed (see Sections 1.2 and 1.3; Chapters 2 to 8).

**Values of interest**

The focus placed on different values associated with nature or nature's benefits to people varies markedly across policy and decision contexts. The IPBES Conceptual Framework (Díaz, 2015) recognises that such values can be of many different types, and this diversity is further described and explored in the draft 'Preliminary guide regarding diverse conceptualisation of multiple values of nature and its benefits, including biodiversity and ecosystem functions and services' prepared by IPBES Deliverable 3d. That guide defines several major types of values of relevance to IPBES activities: instrumental, non-instrumental, anthropogenic, anthropocentric, non-anthropocentric, relational, intrinsic, biophysical, economic and socio-cultural values. It also highlights the importance of future-oriented values associated with nature, and particularly with biodiversity, including bequest, insurance and option values. Any particular policy or decision-making process is likely to focus on a subset, and often a very narrow subset, of all these possible values. Models used to translate exploratory and intervention scenarios into expected consequences for nature and nature's benefits therefore need to be chosen carefully to ensure that response (output) variables projected by these models align well with the values of concern in a given process (see Section 1.2; Chapters 2, 4 and 5). The type and number of values being considered also has implications for the form of assessment or decision-support interface employed – for example whether multiple values need to be combined through multi-criteria analysis or visualisation (see Section 1.4; Chapter 2).

**Drivers of relevance**

The drivers, both indirect and direct, that need to be considered in a given policy or decision-making process will depend partly on the policy goals and options, spatial scale, temporal scale and particular values of nature or nature's benefits being addressed by that process. Some processes may also choose to focus attention on a subset of drivers, or just one particular driver – such as climate change, habitat loss or invasive species – rather than attempting to address all drivers of potential relevance in a given context. This clearly has important implications for the choice of drivers to be projected by scenarios and in turn used as inputs to models translating these scenarios into expected consequences for nature and nature's benefits (see Sections 1.2 and 1.3; Chapters 3, 4 and 5).

**Sectoral breadth of process**

Some policy and decision-making processes will focus exclusively on objectives relating to nature or nature's benefits to people. However, many other processes will consider a broader range of environmental, social and economic objectives, of which only a subset relates directly to nature or its benefits. Such processes are likely to require that the results of any scenario analysis and modelling of nature and nature's benefits are integrated with modelling of other dimensions of human well-being, undertaken across multiple sectors (e.g. health, education or energy) (see Section 1.4; Chapters 2, 3, 5, 6 and 8).

**Governance of process**

Differences in the governance of policy and decision-making processes can also have important implications for the appropriateness of alternative approaches to scenario analysis and decision support. For example, the appropriateness of participatory approaches will depend on the extent to which the policy process is itself participatory, or instead top-down, in nature (see Section 1.4; Chapters 2, 7 and 8).

**Constraints on available time, funding, expertise, knowledge and data**

Finally, all policy and decision-making processes are bound, to varying degrees, by constraints relating to the availability of time, funding and expertise for undertaking associated assessment or decision-support activities, and of knowledge and data to inform these activities. Such constraints can place strong limits on the level of rigour and sophistication that can be achieved in developing and using scenarios and models in any given context, including for example: potential scope to develop new scenarios, as opposed to making use of existing scenarios from previous processes; level of involvement of stakeholders in any such development (e.g. through participatory approaches); and employment of highly integrated process-based modelling techniques, as opposed to simple correlative or expert-based models (see Sections 1.2, 1.3 and 1.4; Chapters 2 to 8).

### 1.5.2 Effective use of scenarios and models in previous assessments and decision-support activities

Scenarios and models of biodiversity and ecosystem services have already been employed effectively in a wide range of assessments informing agenda setting and in decision-support activities informing policy design and implementation. Table 1.1 provides details of selected examples of these applications at global, regional and national scales.

**Table 1.1**: Selected examples of previous applications of scenarios and models of biodiversity and ecosystem services to agenda setting, policy design and implementation at global, regional and national scales.

| | Global Biodiversity Outlook 4 (2014) – GBO4 | IPCC 5th Assessment Report, WG II & III (2014) | Millennium Ecosystem Assessment (2005) – MA | UK National Ecosystem Assessment (2011) – UK NEA | SEA of hydropower on the Mekong mainstream | South African fisheries management |
|---|---|---|---|---|---|---|
| **Spatial scale** | Global | Global | Global | National: United Kingdom (242,000 km²) | Regional: Analysis covers Cambodia, China, Laos, Thailand and Vietnam | National: Coastal fisheries of South Africa |
| **Time horizons** | Present-2020, 2050 | 2050, 2090 | 2050 | 2060 | 2030 | Present-2034 updated every 2-4 years |
| **Position in policy cycle** | Agenda setting, Policy formulation | Agenda setting | Agenda setting | Agenda setting | Policy formulation and implementation | Policy implementation |
| **Authorising environment** | Assessment requested by member countries of the Convention on Biological Diversity (CBD) | Assessment requested by member countries of the IPCC | Initiated by scientific community, then welcomed by UN | Assessment recommended by UK House of Commons as a follow-up to the MA | Strategic Environmental Assessment (SEA) carried out for the Mekong River Commission (MRC) | Evaluation carried out by South African Dept. of Agriculture, Forestry & Fisheries |
| **Issues addressed using scenarios & models** | • Are the Aichi Targets likely to be attained by 2020?<br>• What is needed to achieve the CBD strategic vision for 2050? | How might future climate change impact biodiversity, ecosystems and society? | What are plausible futures of biodiversity and ecosystem services? | What changes might occur in ecosystems, ecosystem services and values of these services over the next 50 years in the UK? | Evaluate social and environmental impacts of dam construction, especially in the main stream of the Mekong river | Implementation of policy on sustainable management of fisheries |
| **Scenarios and models of direct and indirect drivers** | • Statistical extrapolations of trends in drivers up to 2020*<br>• Goal-seeking scenarios & models for analyses up to 2050 ("Rio+20", see Figure SPM.3)<br>• Analysis of wide range of published exploratory and policy screening scenarios at local to global scales | • Emphasis on exploratory scenarios (IPCC SRES)*<br>• Strong focus on models of climate change as direct drivers, some use of associated land-use scenarios*<br>• Some use of goal-seeking scenarios (IPCC RCP)* | • Exploratory scenarios using four storylines*<br>• Models of direct drivers from the IMAGE integrated assessment model* | • Exploratory scenarios using six storylines*<br>- Emphasis on land use and climate change drivers | • Policy screening scenarios using several dam development schemes<br>- Emphasis on economic growth and demand for electricity generation as main indirect drivers<br>• Climate change scenarios also assessed | • Goal-seeking scenarios<br>– Focus on identifying robust pathways for sustainable catch |
| **Models of impacts on nature** | • Statistical extrapolations of trends in biodiversity indicators up to 2020*<br>• Analysis of wide range of published correlative and process-based models<br>- Emphasis on impacts of a broad range of drivers on biodiversity | • Analysis of a wide range of published correlative and process-based models<br>- Emphasis on impacts of climate change on biodiversity and ecosystem functions | • Correlative models (e.g. species-area relationships)<br>- Emphasis on impacts of a broad range of drivers on biodiversity | • Correlative model of species response (birds) to land use<br>• Qualitative evaluation of impacts of land use and climate change on ecosystem functions<br>- Emphasis on habitat change as an indicator of environmental impacts | • Estimates of habitat loss based on dam heights, habitat maps and elevation maps<br>• Estimates of species-level impacts based on dam obstruction of fish migration and on species-habitat relationships | • Population dynamics models of economically important fish<br>• Recently added models of indirectly impacted species (e.g. penguins)<br>• Use of ecosystem-based models under consideration |
| **Models of impacts on nature's benefits** | • Analysis of published studies<br>- Focus on ecosystem services from forests, agricultural systems and marine fisheries<br>- Little evaluation of direct links to biodiversity | • Analysis of wide range of published studies.<br>- Little evaluation of direct links to biodiversity except in marine ecosystems | • Estimates of some ecosystem services (e.g. crop production, fish production) from the IMAGE integrated assessment model | • Qualitative and correlative models of ecosystem services<br>• Focus on correlative methods for estimating monetary value<br>- Emphasis on monetary valuation, except for biodiversity value | • Empirical estimates of fisheries impacts based on reduced migration, and changes in habitat<br>• Diverse methods to estimate changes in water flow & quality, sediment capture, cultural services, etc | • Estimates of total allowable catch (TAC) based on fish population models |
| **Participation of stakeholders** | • Debate and approval by CBD member countries<br>• Dialogs between scientists and CBD secretariat & delegates during assessment process | • Debate and approval by IPCC member countries<br>• Little involvement of stakeholders in scenarios development | • Dialogues with stakeholders during scenario development | • Consultation of stakeholders during scenario development<br>• Adopted by "Living With Environmental Change" partnership of government and non-government stakeholders | • Extensive dialogue involving multiple governments, expert workshops, and public consultations | • Consultation between government, scientists and stakeholders during development of management strategy and setting of TAC |

| | Global Biodiversity Outlook 4 (2014) – GBO4 | IPCC 5th Assessment Report, WG II & III (2014) | Millennium Ecosystem Assessment (2005) – MA | UK National Ecosystem Assessment (2011) – UK NEA | SEA of hydropower on the Mekong mainstream | South African fisheries management |
|---|---|---|---|---|---|---|
| **Decision support tools** | None | None | None | None, but tools under development | Strategic Environmental Assessment methods (see Chapter 2) | Management Strategy Evaluation (MSE, see Chapter 2) |
| | Extrapolations may have contributed to CBD member countries making non-binding commitments in 2014 to increase resources for biodiversity protection | Key documents underlying negotiations of UNFCCC. Commitments of countries to climate mitigation to be discussed Dec 2015 | Increased awareness of the potential for substantial future degradation of biodiversity and ecosystem services | Contributed to Natural Environment White Paper and influenced the development of the biodiversity strategy for England | MRC recommended a ten-year moratorium on mainstream dam construction. One of 11 planned dams is under construction in Laos despite this | Fisheries widely considered to be sustainably managed. Hake fishery is MSC certified |
| **Strengths** | • Novel use of extrapolations for near-term projections<br>• Clear decision context and authorizing environment | • Reliance on common scenarios and models of drivers provides coherence<br>• Clear decision context and authorizing environment | One of the first global scale evaluations of future impacts of global change on biodiversity | Focus on synergies and tradeoffs between ecosystem services and on monetary evaluation | • Clear decision context and authorizing environment<br>• Strong involvement of stakeholders | • Clear decision context & authorizing environment<br>• Policy and management advice clear and updated regularly |
| **Weaknesses** | • Focus on global scale limits applicability to many national and local decision contexts<br>• Lack of common scenarios and models of drivers makes analysis across targets difficult | • Emphasis on climate change, large spatial scales and distant time horizons limits usefulness for policy and management concerning biodiversity and ecosystems | • Very limited set of scenarios and models explored<br>• Decision context unclear and authorizing environment weak | • Heavy reliance on qualitative estimates of impacts of drivers<br>• Biodiversity at species level weakly represented (only birds) | • Highly context specific, especially the empirical models used, and therefore difficult to generalise or extrapolate to larger scales<br>• MRC recommendations non-binding | • Highly context specific<br>• Several key drivers (e.g. climate change) not considered |
| **References** | SCBD (2014), Kok et al. (2014), Leadley et al. (2014), Tittensor et al. (2014) | IPCC AR5 WGII (2014), IPCC AR5 WGIII (2014) | MA (2005) | UK NEA (2011), Watson (2012), Bateman et al. (2013) | ICEM (2010), Chapter 2 of this assessment, ngm. nationalgeographic. com/2015/05/ mekong-dams/ nijhuis-text | Plaganyi et al. (2007), Rademeyer (2014), Chapter 2 |
| **Notes** | *'Methods developed for GBO4* | *'Developed in support of IPCC assessment process* | *'Developed for MA* | *'Developed for UK NEA* | | |

Two contrasting case studies are presented in more detail in Boxes 1.1 and 1.2, illustrating how scenarios and models have been combined effectively to address real-world assessment and decision-support needs at different scales and in different policy contexts.

The first of these (Box 1.1) employs target-seeking (backcasting) scenario analysis, combined with modelling of mean species abundance, to assess development pathways for achieving global sustainability goals. The second study (Box 1.2) was implemented at the watershed scale in Thailand and uses policy-screening scenario analysis to evaluate the consequences of alternative land-use scenarios for the provision of ecosystem services, through the modelling of impacts on water yield and sediment load.

**Box 1.1:** Case study – Rio+20 scenarios

| Project title | Rio+20 scenarios |
|---|---|
| Type of value | Global terrestrial and aquatic biodiversity |
| Driver | Human pressures |
| Temporal extent | Current to 2050 |
| Spatial extent | Global |
| Model use | IMAGE, GLOBIO3 |
| Client | CBD, national governments |

**Multiple challenges, multiple targets**

In 1992, governments worldwide agreed to work towards a more sustainable development that would eradicate poverty, halt climate change and conserve ecosystems. Although progress has been made in some areas, actions have not been able to alter the trends in other critical areas of sustainable development, such as providing access to sufficient food and modern forms of energy, preventing dangerous climate change, conserving biodiversity and controlling air pollution. Without additional effort, these sustainability objectives will not be achieved by 2050.

**Different pathways towards the targets**

To jointly reach the long-term targets on human well-being (eradicating hunger and ensuring full access to modern energy sources), climate change (temperature rise of less than 2°C) and biodiversity conservation (no further loss by 2050), three scenarios were developed. The long-term targets for sustainability were the objective set for 2050 in these target-seeking scenarios (van Vuuren et al., 2012). The three scenarios were based on different strategies of sustainable development, as follows (PBL, 2012):

***Global Technology***: focus on large-scale technologically optimal solutions, such as intensive agriculture and a high level of international coordination, for instance through trade liberalisation;

***Decentralised Solutions***: focus on decentralised solutions, such as local energy production, agriculture that is interwoven with natural corridors, and national policies that regulate equitable access to food;

***Consumption Change:*** focus on changes in human consumption patterns, most notably by limiting meat intake per capita, by ambitious efforts to reduce waste in the agricultural production chain and through the choice of a less energy-intensive lifestyle.

These pathways towards the 2050 targets use different mixtures of policies to enhance productivity and reduce biodiversity loss (Figure Box 1.1), as well as different mixtures to enhance the use of modern energy and reduce climate change.

**Models**

The scenarios were evaluated up to 2050 using the IMAGE 3.0 (Integrated Model to Assess the Global Environment) modelling framework (Stehfest et al., 2014) (http://themasites.pbl.nl/models/image) combined with the GLOBIO 3.0 model (Alkemade et al., 2009) (http://www.globio.info/). IMAGE is an integrated assessment model of global environmental change and enables assessment of the impacts of socio-economic development on the environment, including land use, climate and water flow and pollution. GLOBIO is linked to IMAGE and calculates the impacts of environmental changes on some biodiversity indicators by using cause-effect relationships.

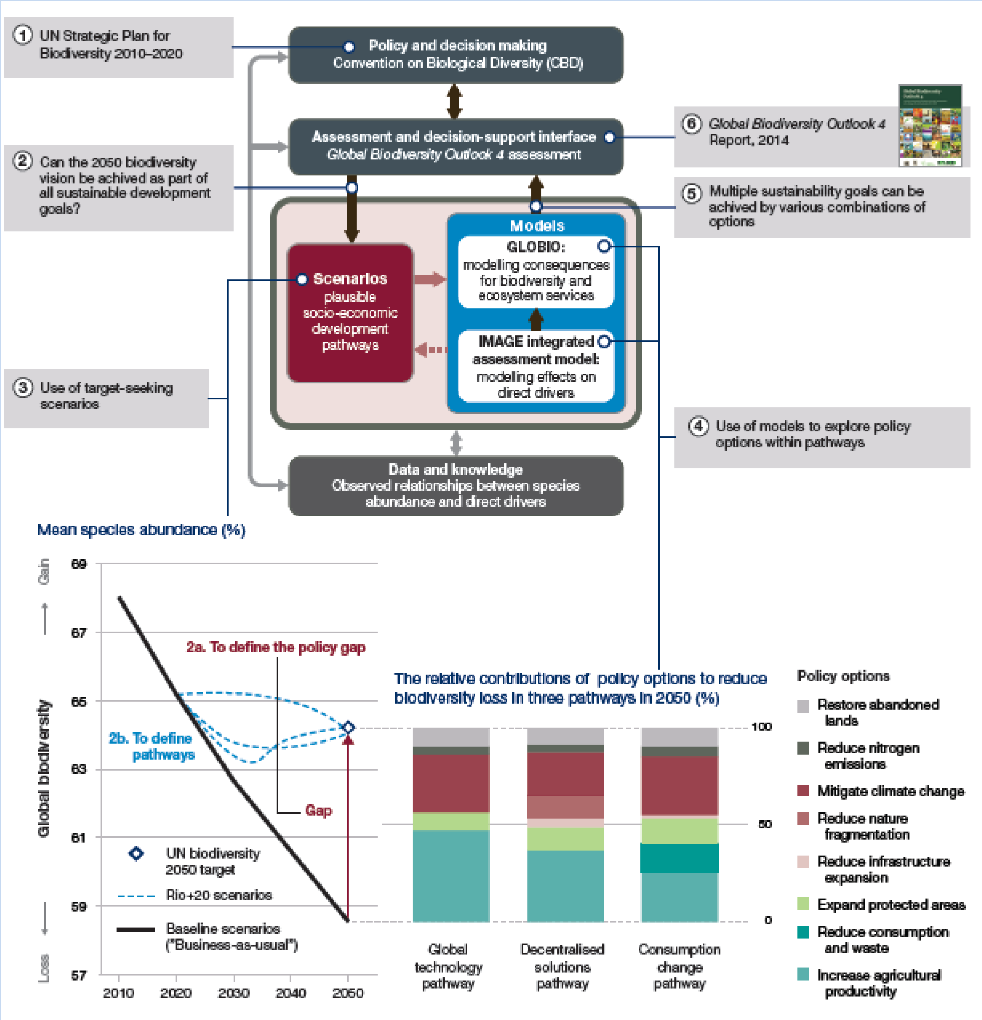



**Figure Box 1.1:** The bottom left-hand graph illustrates the differences between these pathways and a "business-as-usual" scenario in terms of impacts on global biodiversity (as measured by Mean Species Abundance). The right-hand graph indicates the contributions of different components of the three pathways. 'Policy gap' refers to the challenge for policymakers to achieve the goal (PBL, 2012).

The bottom left-hand graph illustrates how these scenarios differ from a "business-as-usual" scenario in terms of impacts on global biodiversity. The bottom right-hand graph shows the relative contributions of indirect drivers to halting biodiversity loss by 2050 compared to the "business-as-usual" scenario. The Global Biodiversity Outlook 4 report was an important factor in discussions at the 12th meeting of the Conference of the Parties to the Convention on Biological Diversity (CBD), which ended with additional commitments to action and funding to achieve the Aichi biodiversity targets.

The results of scenario analyses show that different combinations of policy actions, grouped in the three scenarios, may lead to achieving the multiple sustainability targets. These quantitatively coherent scenarios indicate that eradicating hunger as well as providing full access to modern energy on the one hand, and achieving environmental sustainability on the other, is possible. However, marginal improvements will not suffice; large, transformative changes are needed to realise sustainable development.

**The role of the Rio+20 scenarios in policy support**

Initially a contribution to the Rio+20 conference held in Rio de Janeiro in 2012, the scenarios and their main messages were taken up in the 4th Global Biodiversity Outlook (GBO4) (sCBD, 2014). The parties to the CBD adopted the conclusions of the GBO4 and committed to step up actions to achieve the Aichi biodiversity targets, including a pledge by national governments to double funding for necessary actions (CBD, http://www.cbd.int/doc/press/2014/pr-2014-10-17-cop-12-en.pdf). Additional initiatives were launched to enhance the biodiversity perspective in sustainable commodity production (CBD, http://www.cbd.int/doc/press/2014/pr-2010-10-16-commodities-en.pdf). The outcomes from the scenario analyses provided underlying arguments for these decisions and initiatives.

**Box 1.2:** Case study – Thadee watershed, Thailand

| Project title | Thadee watershed, Thailand |
|---|---|
| Type of value | Watershed services |
| Driver | Land-use change |
| Temporal extent | 2009-2020 |
| Spatial extent | Catchment (112 km$^2$) |
| Model use | CLUEs, InVEST, RIOS |
| Client | Local stakeholders, local government |

The Thadee watershed located in southern Thailand covers approximately 112 km$^2$. Water from the watershed is mainly used for agriculture by upstream farmers and household consumption by downstream people in the Nakhon Srithammarat municipality. However, natural forests in the watershed have been degraded and transformed to monocultures (fruit trees and rubber plantations) due to a governmental subsidy programme. The ECO-BEST project, co-funded by the EU, German government (GIZ) and Thailand (Department of National Parks, Wildlife and Plant Conservation and Kasetsart University), worked with scientists to quantify water yield and sediment load according to different land-use and rainfall scenarios between 2009 and 2020 (Trisurat, 2013). The CLUE-s (Conversion of Land Use and its Effects) model (Verburg and Overmars, 2009) was used to allocate future land demands based on two scenarios – agriculture development and conservation. In addition, InVEST (Integrated valuation of ecosystem services and trade-offs) (Nelson et al., 2009) and USLE (Universal Soil Loss Equation) models were employed to estimate water yield and soil erosion respectively. The modelling results clearly show that intensifying land-use change due to the rapid expansion of rubber plantations and extreme rainfall will generate a high risk of major sediment loadings and overland water flows due to the force of rainfall and decreased evapotranspiration from vegetation. Applying the economic model RIOS (Resource Investment Optimization System) (RIOS, Vogl et al. (2013)), the project team together with stakeholders could identify which conservation activities (e.g. protection, reforestation and the promotion of mixed-cropping systems) should be implemented – and where – to yield the highest return on investments and to enhance watershed services. The municipality has agreed in principle to find the best practical mechanism for collecting payments from tap water clients and downstream ('payment for watershed services') to implement the above activities.

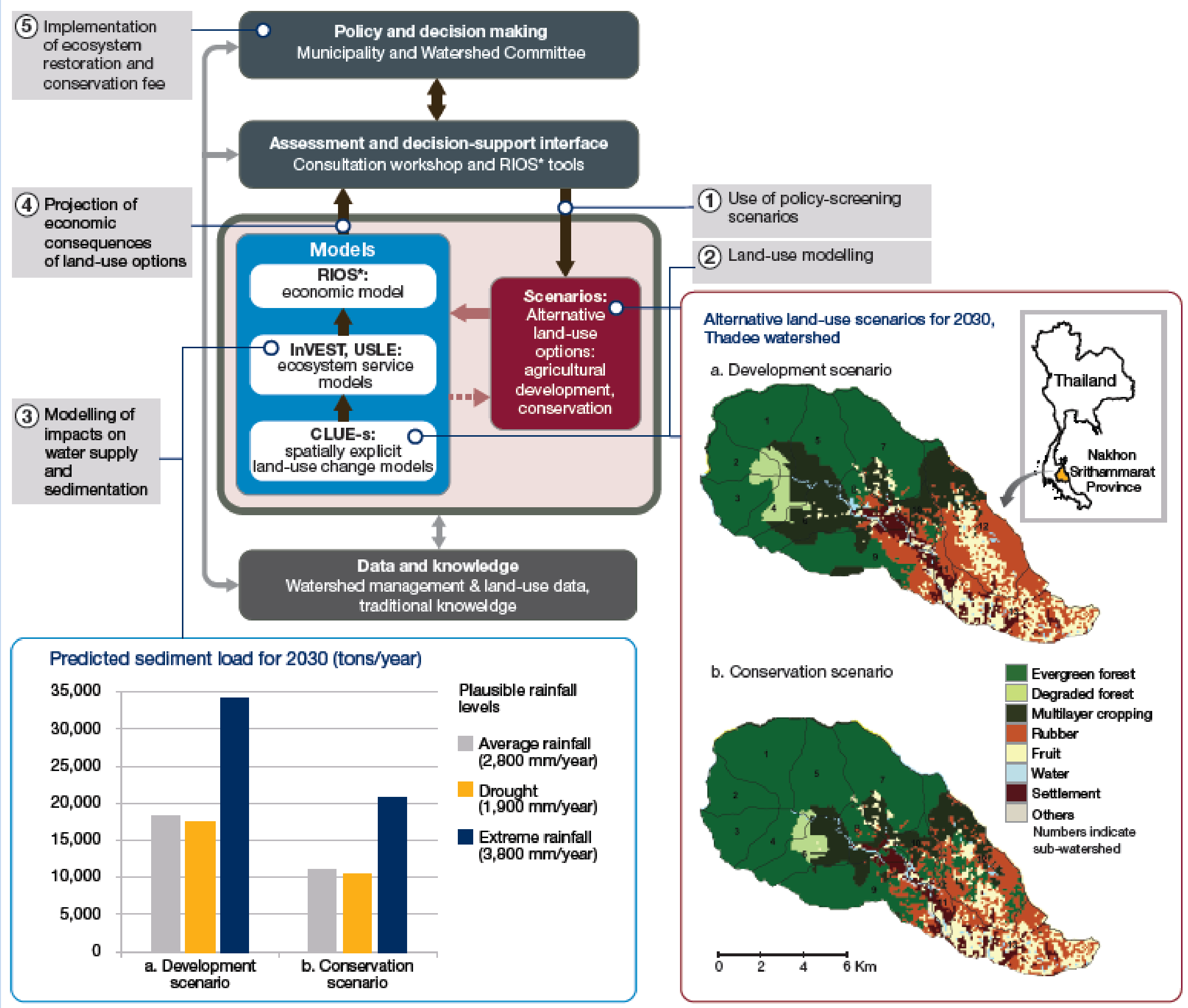



**Figure Box 1.2**: Results from an integrated scenarios and modelling assessment of ecosystem services for the Thadee watershed in Nakhon Srithammarat Province, Thailand. The right figure shows the expansion of agriculture under the agricultural development scenario, as compared to the conservation scenario. The bottom-left figure shows that sediment load to the river depends on the amount of rainfall and that sediment load is much less under the conservation scenario (Trisurat, 2013).

## 1.6 Recognising and addressing current limitations of scenarios and models

Previous sections of this chapter have outlined the many ways in which scenarios and models of biodiversity and ecosystem services can contribute significantly across all phases of the policy cycle. But what are the challenges that need to be overcome to achieve the broader application of these approaches? Identifying these challenges, and suggesting effective means of overcoming them, are themes that run through all the chapters of this report. Some of the most important challenges relate to a general lack of understanding among policy and decision-making practitioners regarding the benefits of using scenarios and models (see Chapter 2), and a shortage of the human and technical resources needed to enable this use in many parts of the world (see Chapter 7). Various forms of capacity building that could be used to address challenges of this type are described in Chapter 7. Other challenges are more technical in nature and concern limitations in currently available scenarios and models. While these limitations are also examined in depth throughout the remainder of the report (Chapters 2 to 6 and 8), three issues cutting across this discussion warrant introduction at this point.

### 1.6.1 Gaps in the focus and coverage of available scenarios and models

Section 1.5.1 stressed the importance of matching the types and characteristics of scenarios and models employed in any given policy or decision-making process to the needs of that process. Different processes often require very different types of scenarios and models, operating at different spatial and temporal scales, focusing on different ecosystems, addressing different sets of drivers and, in the case of models, projecting changes relating to different values of nature or nature's benefits (Figure 1.7).

Significant gaps currently exist in the availability of scenarios and models, and in methods for their derivation, to serve existing and emerging needs across the full range of policy and decision contexts.

Published studies of scenarios and models (as accessed for the graph presented in Figure 1.1) show a strong bias towards terrestrial ecosystems and towards climate change as the driver of interest. Nearer-term drivers such as habitat loss and modification, invasive species, pollution and overexploitation have received insufficient attention (FRB, 2013). Marine ecosystems are reasonably well represented, with many studies focusing on fisheries management or climate-change impacts on marine biodiversity and ecosystems (e.g. Dunstan et al., 2011; Sumaila et al., 2011). However, freshwater ecosystems are under-represented in existing analyses compared with terrestrial ecosystems. Biodiversity models are heavily biased towards the species level followed by community-level studies, with relatively few models addressing the genetic level. Animals and plants are represented roughly equally, but micro-organisms are infrequently addressed. There is also a strong bias of scenarios and models towards mid- and end-21$^{st}$ century outcomes (FRB, 2013), whereas many managers and policymakers are more focused on nearer-term goals (e.g. Aichi biodiversity targets for 2020, sCBD, 2014). Comparisons between modelled outcomes in the past and observations are also rare, even though these could strengthen confidence in future projections. Spatial scales of scenarios and models employed in assessments typically focus on national to global scales. Few assessments account for the vast amount of information from scenarios and models applied at the sub-national scale, which is a more pertinent spatial scale for many decision-making processes. Finally, in relation to ecosystem services, the scenarios and models employed in most assessments have rarely dealt with services outside of food production and carbon storage (but see UK NEA, 2011; PBL, 2012), even though other types of ecosystem services are often key considerations in decision making.

### 1.6.2 Deficiencies in underpinning knowledge and data

Most models build on established knowledge and data to describe relationships of interest. Data are used to guide the design of models, calibrate model parameters and validate predicted outcomes.

The effectiveness of scenario analysis and modelling in informing policy and decision making depends on the relevance, quality, quantity and availability of data and knowledge (scientific, indigenous and local). Modelling does not replace the need for good data and knowledge, but instead provides a means of extracting maximum value from the best-available information at any point in time.

The quality of modelled outputs for use in assessments and decision support will always be constrained by the quality and quantity of the underpinning information. The importance of linking future applications of scenario analysis and modelling with ongoing efforts and initiatives around gap-filling data collection and knowledge acquisition is addressed in depth in Chapter 8. The importance placed by

IPBES on this issue is also reflected by the establishment of two key activities under the IPBES Work Programme: the Task Force on Knowledge and Data Generation; and the Task Force on Indigenous and Local Knowledge (ILK).

This methodological assessment includes particular consideration (in Chapters 5 and 7) of the contribution that indigenous and local knowledge can make to filling information gaps, and to enabling the successful application of scenarios and models to policy and decision making, including through the use of participatory approaches to scenario and model development. For example, the mobilisation of ILK through participatory approaches can help to ensure that indigenous peoples have an integral and meaningful role in making decisions and in contributing to natural resource management that affects their future, either directly or indirectly (Emery, 2000). In terms of scenarios and models, this knowledge is crucial in order to accommodate fundamental aspects of day-to-day life and cultural complexes that also encompass language, systems of classification, resource-use practices, social interactions, ritual and spirituality. Combining ILK with scientific knowledge will, in many cases, lead to greater benefits than can be achieved by treating these knowledge sources separately (Thaman et al., 2013).

### 1.6.3 Challenges in dealing with uncertainty

The term ‘uncertainty’ appears repeatedly throughout the remaining chapters of this assessment report. To properly appreciate the importance, and varied implications, of this issue for the discussion and use of scenarios and models, it is vital to first recognise that uncertainty can take a diversity of forms, arising from very different sources. Various typologies of uncertainty have been proposed in the environmental sciences literature (e.g. Regan et al., 2002; Skinner et al., 2014). For the purposes of this report, four major sources of uncertainty are recognised:

- **Linguistic uncertainty** – imprecise meaning of words, including vagueness and ambiguity;
- **Decision uncertainty** – variation in subjective human judgments, preferences, beliefs and world views;
- **Stochastic uncertainty** (also known as ‘aleatoric uncertainty’) – the random behaviour or unpredictability of complex natural, social and economic systems, particularly in relation to future states;
- **Scientific uncertainty** (also known as ‘epistemic uncertainty’) – imperfect knowledge or data on the system being described.

Each of these sources of uncertainty has particular implications for the description and use of scenarios and models. Throughout this report, linguistic uncertainty is addressed largely through the careful definition of terms, including in the report’s glossary. Previous sections of this chapter have already introduced strategies for dealing with decision uncertainty, for example by ensuring that employed scenarios and models are well matched to different policy and decision contexts, and that assessment and decision-support interfaces enable the effective analysis of synergies and trade-offs between multiple values and objectives.

Stochastic uncertainty is the very challenge that exploratory scenarios are designed to address. The use of exploratory scenarios accepts that future trajectories of drivers of change in nature and nature’s benefits will depend on events and actions that are yet to occur, and that are highly unpredictable. This uncertainty is therefore accommodated through the construction of a set of plausible futures rather than a single future (see Chapter 3).

The purpose of exploratory scenarios is not to reduce stochastic uncertainty (which, by definition, cannot be reduced), but rather to convey realistic estimates of this source of uncertainty to policy and decision making (Enserink et al., 2013).

Scientific uncertainty associated with models used to translate scenarios into expected consequences for nature and nature's benefits needs to be minimised as much as possible. However, all models have limitations, and no model can generate perfect predictions. It is therefore highly desirable that levels of scientific uncertainty associated with model outputs are estimated, and accounted for effectively in decision making (see Chapter 2).

Scientific uncertainty is an unavoidable outcome of the very nature of models being simplifications of reality and condensations of current knowledge. In the remainder of this report, many shortcomings and gaps in models will be addressed. The most important of these gaps relate to deficiencies in knowledge about key variables and relationships; loss of information when simplifying complex real-world systems to models; uncertainty in estimating the values of parameters and variables; lack of sufficient data of the right quality to validate models; and error propagation, especially within complex models.

## 1.7 Structure of this report

Methods for modelling different components of socio-ecological systems (i.e. elements of the IPBES Conceptual Framework) are increasingly being integrated within a single modelling framework (e.g. through so-called 'Integrated Assessment Models'). Likewise, the boundary between methods for modelling and methods for scenario development, assessment and decision making is becoming increasingly fuzzy as a result of the closer coupling of approaches across these domains. However, in the interests of breaking the overall challenge down into manageable pieces, Chapters 2 to 5 each focus on a particular aspect or component of this challenge (Figure 1.8). Linkages and dependencies between these topics, and the need for any given application of scenarios and models to consider these issues together, rather than sequentially, are emphasised throughout.

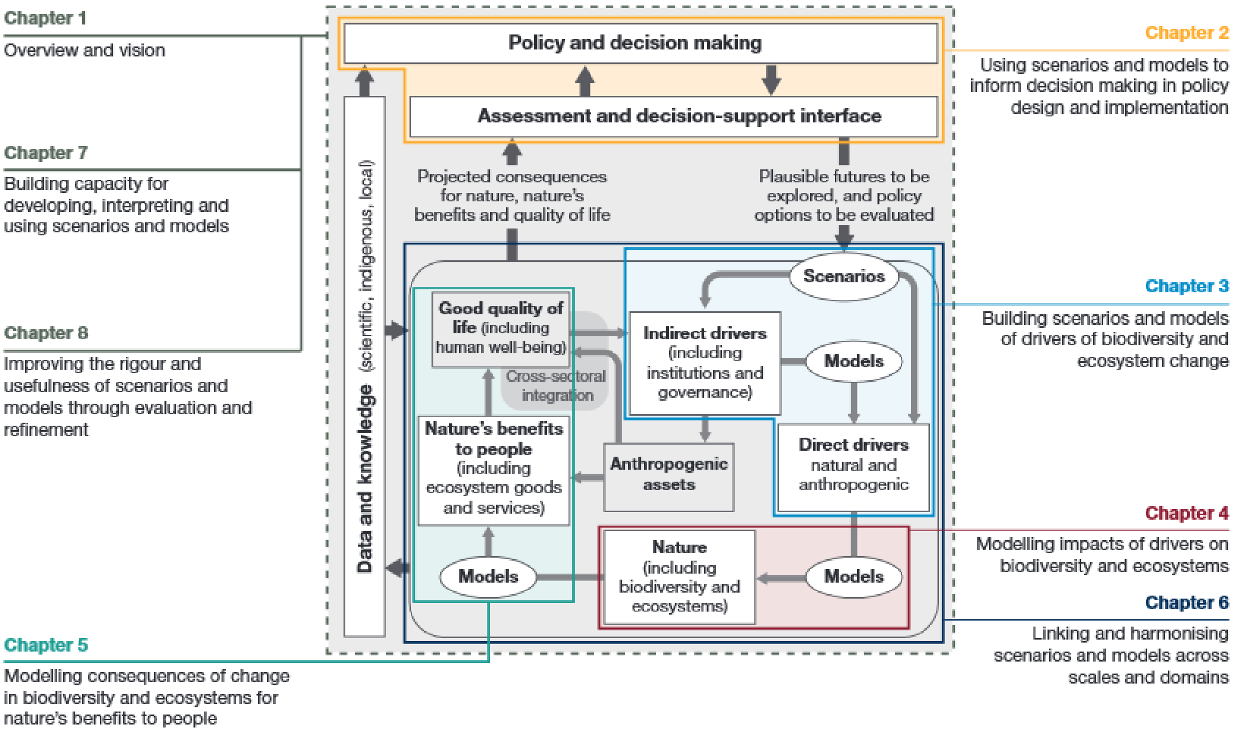



**Figure 1.8:** Relationship of chapters to the components depicted in Figure 1.3.

Chapter 2 examines issues around 'using scenarios and models to inform decision making in diverse policy, planning and management contexts'. It provides an overview of policy, planning and management contexts in which scenarios and models can aid assessment and decision making, and considers lessons learnt from established decision-support paradigms and frameworks that make strong use of scenarios and models. Particular emphasis is placed on the importance of aligning the design of scenarios and models with the particular needs of assessment and decision-making processes associated with different phases of the policy cycle, and of dealing with uncertainty in scenarios and models employed in decision making.

Chapter 3 addresses challenges associated with 'building scenarios and models of indirect and direct drivers of change in biodiversity and ecosystems' to address the assessment and decision-making needs identified in Chapter 2, and presents a typology of exploratory and intervention scenario sub-classes linked to major phases of the policy cycle. It reviews approaches to developing plausible scenarios of indirect drivers and lessons learnt from the previous development and application of such scenarios in assessments at global and regional scales. It then reviews methods for modelling expected consequences of indirect-driver scenarios for direct drivers of change in biodiversity and ecosystems across terrestrial, freshwater and marine systems (as input to models of biodiversity and ecosystem responses considered in Chapter 4).

Chapter 4 deals with 'modelling impacts of drivers on biodiversity and ecosystem properties and processes'. It explores existing and emerging approaches (both correlative and process-based) to modelling impacts of a broad range of direct drivers (from Chapter 3) on biodiversity across multiple levels (e.g. population, species and community) and dimensions (e.g. composition, structure and function) of biological organisation, and ecosystem properties and processes (e.g. biomass and primary production).

Chapter 5 focuses on 'modelling consequences of change in biodiversity and ecosystems for nature's benefits to people'. It explores challenges associated with translating modelled biophysical changes in biodiversity and ecosystem properties and processes (from Chapter 4) into expected consequences for benefits to people (including ecosystem services), human well-being and good quality of life. It emphasises the importance of recognising that different decision-making processes may require careful consideration of differences in the values that people involved in these processes place on, or derive from, nature.

The remaining chapters of the report explore, in greater depth, three particularly important cross-cutting challenges facing the ongoing development and application of scenario analysis and modelling from an IPBES perspective (Figure 1.8).

Chapter 6 articulates the need for better 'linking and harmonising scenarios and models across scales and domains' and proposes practical strategies and solutions for achieving this in both the short and longer term. These include approaches to more closely linking and harmonising scenarios and models across different scales of assessment and decision making, and to achieving the closer coupling of scenarios dealing with different drivers and models focusing on different dimensions or levels of biodiversity or on different ecosystem functions or services (as covered separately in Chapters 3, 4 and

5).

Chapter 7 addresses the challenge of 'building capacity for developing, interpreting and using scenarios and models' by proposing practical strategies that account for regional and cultural diversity in perspectives on, and capacity for, scenario analysis and modelling. These include approaches to improving regional and national access to, and training in, appropriate data sets and software tools; developing methods for better incorporating local data and knowledge; and developing effective strategies for mainstreaming scenarios and models into assessment and decision-making processes across scales and across different policy, planning and management contexts.

Chapter 8 adopts a forward-looking perspective in addressing the challenge of 'improving the rigour and usefulness of scenarios and models through ongoing evaluation and refinement'. It lays out a comprehensive vision and strategy for taking scenario analysis and modelling of biodiversity and ecosystem services to a whole new level of rigour, credibility and utility by more closely linking this field to parallel initiatives in biodiversity/ecosystem data acquisition and thereby establishing a rigorous foundation for ongoing model evaluation and calibration, and advancing the fundamental science underpinning the development and application of scenarios and models through carefully prioritised research activities.

Each of the chapters includes a set of 'Key findings' and 'Key recommendations' at the start of the chapter. Key findings are general messages that arise from the critical analyses in this assessment and are aimed at a broad audience. Key recommendations are based on the key findings and more specifically address IPBES and experts involved in its deliverables. The key recommendations provide explanations of a wide range of actions that could be undertaken or stimulated by IPBES.

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# 2 Using scenarios and models to inform decision making in policy design and implementation

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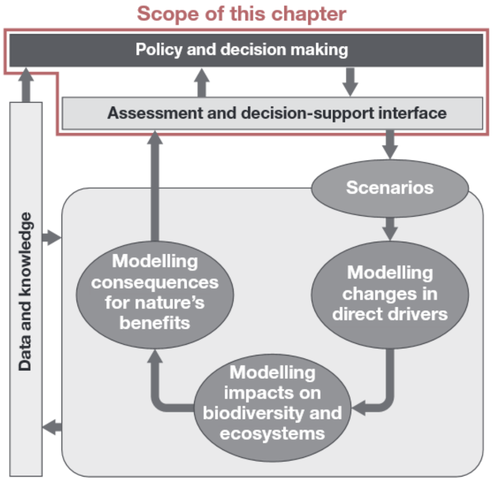


**Purpose of this chapter:** Provides an overview and typology of policy and decision-making contexts; sets the scene for Chapters 3, 4 and 5 to identify the scenarios and models needed in these different contexts; and critically reviews major decision-support approaches for interfacing scenarios and models with policy and decision making.

**Target audience:** A broader, less technical audience for the overview of policy and decision-making contexts but a more technical audience for the review of particular decision-support approaches.

# Key findings

**The decision context determines the most appropriate decision-support tool for any situation. Decision context can be defined in terms of multiple attributes such as cultural and ecological complexity, temporal scale and complexity of governance. A multitude of decision-support tools and approaches exist that can be utilised at the decision-support interface to integrate information, address divergent stakeholder objectives and beliefs, and help deal with the many challenges and complexities facing decision makers. For every decision context, there are several decision-support approaches and tools that may be appropriate. Decision-support tools include scenarios, models of biodiversity and ecosystem services, and decision-making protocols, frameworks and approaches such as multi-criteria decision analysis, numerical optimisation and integrative frameworks such as management strategy evaluation and structured decision making.** Scenarios, models and decision-support frameworks and protocols are used to help set the policy agenda and support policy design, implementation and review. However, their influence on decisions is not always well documented.

**Only a small proportion of decisions that impact on biodiversity and ecosystem services are explicitly considered environmental decisions, and a very low proportion of such decisions utilise scenarios, biodiversity and ecosystem services models and decision frameworks and approaches. Barriers to the use of decision-support tools in environmental policy agenda setting, design and implementation range from a lack of appreciation among decision makers about the potential benefits of using models and scenarios, to a lack of willingness on the part of some modellers to properly engage in real-world decision making and undertake relevant analyses.** Of the case studies reviewed that successfully applied decision-support tools, the dedication and continuity of facilitators and modellers in close collaboration with decision makers was a consistent feature throughout the decision-making processes. Primary impediments to the widespread use of models and scenarios in decision making include: a general lack of trust in modellers, models and scenarios; a lack of understanding and technical knowledge among decision makers to allow them to understand outputs and appreciate the positive role that models and scenarios can play; a general lack of decision-support, modelling and scenario analysis skills relative to the number of policy design and implementation challenges; a lack of data to underpin the models and scenarios of most interest to policymakers and managers; a lack of willingness on the part of some modellers to engage fully in real-world decision problems and develop and communicate in a non-technical way the most relevant scenarios and models for the problem at hand; a lack of willingness of modellers to engage in participatory processes involving other knowledge traditions and the translation of model outcomes to other knowledge traditions; a lack of transparency in approaches to modelling and scenario development; and complex political agendas that are not amenable to the transparency ideally associated with good modelling and scenario analysis.

**There is often a mismatch between the spatial and temporal grain and extent of biodiversity and ecosystem services models and the policy design and implementation needs of decision makers. The cross-scale, cross-sectoral and cross-ecosystem linkages necessary for decision makers and stakeholders to understand more fully the implications of decisions are often absent. While significant progress has been achieved in understanding impacts and feedbacks between environmental variables across spatial scales, the needs of policy and decision makers are rarely paramount in determining data needs, necessary model outputs, and the types of scenarios and models that are developed.** Knowledge about the state of key biodiversity and ecosystem service variables and how socio-ecological systems function and respond to stressors and human interventions depends on collecting new data at multiple organisational levels and monitoring the impacts of decisions. Decisions will be best supported if assessment and decision-support needs drive data collection priorities and the choice of scenarios, models and model outputs.

**There are very few agreed standards of best practice for some of the most important and widely used assessment and decision-support tools, such as strategic environmental assessment. As a consequence, many assessments default to the lowest common denominator, especially when it comes to assessing the impacts of large, complex development proposals on biodiversity and ecosystem services.** There is an opportunity for IPBES to raise the bar on such assessments by promoting standards of best practice in assessment and decision support that require state-of-the-art scenario analysis and modelling approaches be coupled with integrative, participatory decision-support protocols and frameworks.

**Uncertainty may contribute to poor decisions with negative social, economic and environmental outcomes. Decision-making processes are most likely to be effective if important uncertainties are characterised and addressed in policy, planning and management.** Environmental problems and the process of finding technical and management solutions to these are challenged by stochastic, linguistic, scientific and decision uncertainties with various levels of complexity and reducibility. Technical approaches to analysing the impacts of uncertainties on decision outcomes, including analysing the robustness of decision or planning options to various uncertainties, can provide useful information to decision makers. Socially acceptable trade-offs under uncertainty can also be achieved through deliberation that allows feedback and learning among decision makers and stakeholders.

**Examples of the integration of indigenous and local knowledge systems in models and scenarios and improved decision outcomes through the participation of indigenous and local people are rare, although encouraging examples can be found.** Ecological systems are complex and difficult to interpret with only one scientific discipline or knowledge tradition. The livelihoods of traditional knowledge holders are highly dependent on biodiversity and ecosystem services, but these people are frequently explicitly and implicitly excluded from policy decisions, particularly at and above the national level. In order to make better use of indigenous and local knowledge systems and encourage greater participation, efforts must be made to enhance capacity of indigenous and local peoples to allow them to participate in decision-making fora and to understand, interpret and contribute to modelling and scenario development.

## Key recommendations

**IPBES global and regional assessments can be an important forum for fostering stronger links between ecosystem services and biodiversity experts, social scientists, modellers, decision-support experts, decision makers, stakeholders and indigenous and local peoples.** This can be achieved by allowing global and regional assessments to go beyond biophysical and socio-ecological assessments of states and trends to become fora in which policy options are expertly evaluated using a broad range of relevant data, models, scenarios and policy-evaluation (decision-support) methods and approaches. Increased collaboration between modellers and decision makers will lead to increased trust, better and more relevant models and scenarios, and a culture of decision support based on models and scenarios suited to complex policy and political agendas.

**The typology and evaluations presented in this chapter provide a preliminary guide to which types of decision-support frameworks, protocols and approaches are relevant to any particular policy design, implementation and review context.** When considering which decision-support frameworks, protocols and approaches are most relevant to a policy design, evaluation or implementation problem, IPBES deliverables (especially Deliverables 2b, 2c, 3b and 4c) could benefit from using the decision-context typology and the decision-support tools strengths and weaknesses evaluation presented in this chapter.

**The IPBES Task Force on Capacity Building (Deliverables 1a/1b) could build on this assessment by seeking to foster and develop capacity in decision-support expertise – including skills in biodiversity and ecosystem services modelling, scenario development and analysis – and improved understanding of and expertise in the process of policy evaluation and decision support. Policy evaluation and** decision-support processes should utilise a variety of tools, protocols and frameworks such as multi-criteria decision analysis, optimisation, structured decision making and other approaches that are summarised and reviewed in this chapter.

**Outside of IPBES assessments, IPBES could promote fora and networks that link ecosystem services and biodiversity experts, social scientists, modellers, decision-support experts, indigenous and local peoples, stakeholders, and decision makers.** The Task Force on Capacity Building (Deliverables 1a/1b) and the policy and decision tools catalogue (Deliverable 4c) could use the decision-context typology and the evaluation of decision-support tools strengths and weaknesses presented in this chapter to help ensure that modelling and scenario analysis tools recommended to decision makers and their stakeholders are appropriate to their policy and decision context.

**The IPBES Task Force on Knowledge, Information and Data, in combination with funding agencies and data providers, could promote and facilitate data collection targeted towards decision-making needs and supporting the monitoring of the impacts of decisions on the composition, structure and function of biodiversity and ecosystem services.** IPBES global, regional and thematic assessments have the opportunity to identify data collection priorities that best address decision makers' needs by engaging decision makers, indigenous and local peoples, and stakeholders in IPBES assessments and by utilising the decision-support frameworks, approaches and tools described in this chapter to prioritise data gap filling.

**The IPBES deliverable on policy and decision tools (Deliverable 4c) and the scenarios and models expert group (Deliverable 3c) could promote standards of best practice in assessment and decision support that require state-of-the-art scenario and modelling approaches be coupled with integrative, participatory decision-support protocols and frameworks when undertaking assessments of policies, plans and programmes that impact on biodiversity and ecosystem services.** This can be achieved through the establishment of networks of decision-making practitioners, modellers and experts in biodiversity and ecosystem services with the explicit aim of raising the bar on current approaches to the assessment of policies, plans and programmes.

**Thematic, regional and global assessments could identify capacity needs for dealing with scientific uncertainties during decision making and work with the Task Force on Capacity Building to foster and facilitate improved capacity for characterising, communicating and dealing with uncertainties that impact on decisions in a way that is consistent and based on agreed standards.** IPBES assessments should seek to identify the uncertainties that impact most heavily on the capacity of decision makers to make decisions that are beneficial to biodiversity and ecosystem services. This will require discriminating between uncertainties that are relatively benign, and uncertainties that are important because they impair decision making.

**Thematic, regional and global assessments, in cooperation with the IPBES Task Force on Indigenous and Local Knowledge (Deliverable 1c), could use assessment and policy-support approaches that integrate multiple spatial and temporal scales and recognise the importance of multiple and diverse knowledge systems. Formal participatory mechanisms need to be established to ensure local and indigenous participation and the effective exchange of information between scientists and local and indigenous peoples.**

# 2.1 Introduction

Decision-support protocols have advantages over unaided decision making because they provide and document the logic behind decisions. Apart from buffering against cognitive limitations and negative group dynamics, a documented and traceable decision-support protocol will encourage decision makers to be clear about judgments and assumptions (Bedford and Cooke, 2001). Scenarios and models can play several important roles within decision-making processes, including: i) setting a policy agenda by highlighting previously poorly-documented threats or opportunities; ii) transparently representing assumptions about cause-effect pathways that link policies and actions to outcomes; iii) reducing complexity by synthesising, analysing and representing multiple sources of information and evidence in a way that is most appropriate for the decision at hand; iv) exploring and identifying unforeseen consequences of policies and actions; and v) providing a means to synthesise and interpret policy, planning and management evaluation information, including monitoring data.

## 2.1.1 The policy cycle, knowledge needs and the role of assessment

An extensive literature documents policy theory and practice and processes that influence policy design and implementation (e.g. Sabatier and Weible, 2014).

While there are many competing models describing policy processes, the simplicity and communication value of the four-phase policy cycle (Howlett et al., 2009) is of value here in providing a context for discussion about decision-support tools relevant to decisions that impact on biodiversity and ecosystem services. Under this model, decision making occurs in four phases of the policy cycle: agenda setting (including problem identification), policy design, policy implementation and policy review (Figure 2.1).

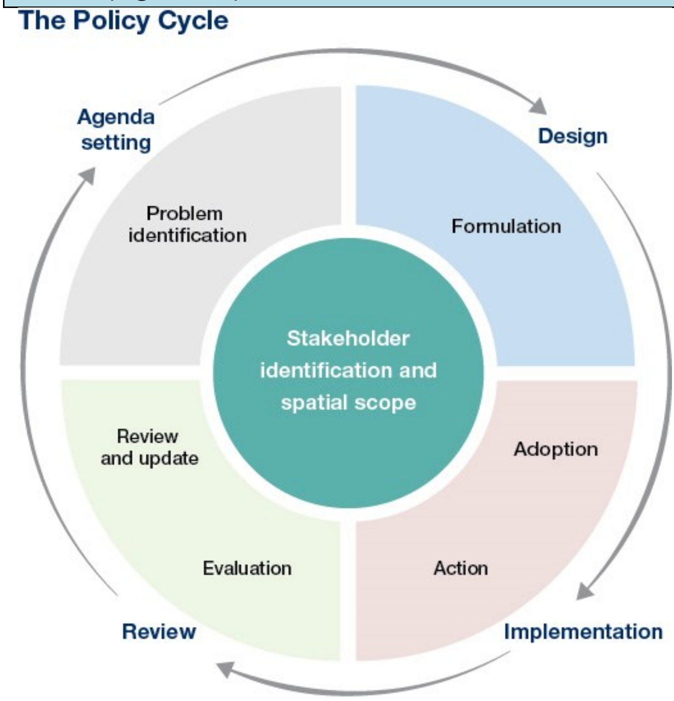


**Figure 2.1:** A theoretical framework for agenda setting, policy design, implementation and review (modified from Howlett et al. (2009). Although empirical evidence shows that real-world decision making does not usually follow an idealised sequence of discrete stages (Jann and Wegrich, 2007), the policy cycle helps organise the discussion of the role of scenarios, models and decision-support approaches in decision making that occurs in subsequent chapters. Numerous published frameworks exist that describe similar steps and approaches for structuring and implementing policy and decision making under uncertainty and complexity, including adaptive management and adaptive planning approaches (McFadden et al., 2011; Walters, 1986).

The four phases of the policy cycle have specific ***knowledge needs*** that can be partly met by biodiversity and ecosystem service models implemented under scenarios exploring the implications of policy settings (Figure 1.3 in Chapter 1). For example, problem identification and problem scoping, including the identification of the scope of assessments and stakeholders, are all activities that take place under the broad banner of agenda setting. In many situations, the modelling of direct and indirect drivers of biodiversity and ecosystem services, embedded in exploratory scenarios (Chapters 3 and 5), can provide important insights into the nature and magnitude of problems and opportunities that drive the development of specific policy options. This type of exploration can trigger new policy agendas. For example, Section 2.3.1 describes how a series of agenda-setting scenario analyses starting with the first Global Biodiversity Outlook (GBO) (sCBD, 2001) contributed to the development of, and agreement on, the Aichi biodiversity targets (MA, 2005; Alkemade et al., 2009; Leadley et al., 2014). Similarly, the policy design, implementation and review phases have knowledge needs that can be partly met through the use of scenarios, models and decision-support methodologies. In policy implementation, for example in land-use planning, scenarios, models and other formal decision-support approaches are often used to help identify which activities will be allowed or encouraged in particular parts of the landscape in order to achieve landscape-level objectives for a range of criteria such as agricultural productivity, tourism service provision and biodiversity conservation (FAO, 1993; SAPM, 2009). Policy implementation often involves management decision making in the face of uncertain benefits and costs due to complex ecological or social system dynamics, and multiple criteria for measuring success. In such cases, decision support – including scenarios, models and structured approaches for analysing trade-offs – can be extremely useful for ensuring that management is transparent, effective and efficient in meeting objectives for biodiversity, ecosystem services and other criteria (Runge et al., 2011b).

During policy review, the outcomes of previously adopted policies can be compared to hypothetical counterfactual or alternative scenarios (Chapter 3, Table 3.1). Scenarios and models can be used to estimate biodiversity or ecosystem service outcomes under hypothetical policy settings alternative to the ones actually implemented. This sort of analysis is often called post hoc or ex-post evaluation, and can provide valuable information about how to adjust policy settings with the aim to better achieve desired outcomes in the future, or simply as a form of transparent reporting on the performance of policies or programmes. For example, Joppa and Pfaff, (2010) reviews a statistical technique called 'matching' to compare observed forest conservation status against counterfactual scenarios of forest loss in the absence of protection to estimate the effectiveness of forest conservation (Chapter 3).

This chapter sets the assessment and decision-making scene for the three other chapters of this deliverable that provide more detail on scenario development and modelling approaches relevant to particular decision contexts (Chapters 3, 4 and 5).

It links to Chapter 3 by identifying types of scenarios required to underpin decision making, and to Chapters 4 and 5 by identifying the role of biodiversity and ecosystem service model outputs in agenda setting, policy design, implementation and review. This chapter also provides the foundation for Chapters 6 and 7 by highlighting the scales and domains over which different types of decisions occur, and the capacity-building needs in the area of model-supported decision analysis. A view to the future of agenda setting and decision making offers an entree to Chapter 8 by highlighting future developments that may see the increased use of scenarios and models in decision making.

### 2.1.2 Aims and audience

This chapter aims to inform readers about the possibilities and opportunities for using scenarios, models and decision-support protocols to support decisions in each phase of the policy cycle, from agenda setting to policy design, implementation and review. A decision-context typology is provided that defines the range of decision contexts in which scenarios and models may be useful. Decisions that impact on biodiversity and ecosystem services are defined according to ***decision-context attributes***. The aim is to try and reduce some of the complexity and confusion about the range of tools and decision protocols that may help to support decisions that impact on biodiversity and ecosystem services. The chapter seeks to improve understanding about the contexts in which decision-support approaches may be useful, and demonstrate how they may be enhanced with the use of scenarios and models. Examples of where decision-support approaches have been successfully integrated with scenarios and models to improve decisions are described.

This chapter principally addresses the following activities within the IPBES work programme: regional and global assessments (Deliverables 2b/2c), thematic assessments (Deliverable 3b), the scenarios and models expert group (Deliverable 3c) and the deliverable on policy and decision tools (Deliverable 4c), which will develop an online catalogue of policy-support tools and methodologies relevant to IPBES-related activities. Findings are also relevant to Deliverables 1a/1b on capacity building, Deliverable 1c on indigenous and local knowledge, Deliverables 1d/4b on knowledge information and data, Deliverable 3d on valuation and Deliverable 4d on stakeholder mapping and engagement.

## 2.2 Decision-making context

### 2.2.1 Attributes that define decision context

Almost every policy, plan and action in every sector from health to manufacturing, and at every spatial and organisational scale from the individual to the global, impacts in some way on biodiversity and ecosystem services.

The number and types of decisions made appear to defy classification and are practically infinite (Fisher et al., 2009). The bulk of decisions or choices made on a daily basis that impact on biodiversity and ecosystem services are seldom described or conceived of as *environmental* decisions (a decision in which environmental considerations are explicit). Almost all are undertaken by people outside the environmental sector with little or no consultation with environmental professionals. The following paragraphs describe attributes of the decision context (Table 2.1), with a focus on decisions that are readily identified as 'environmental decisions'.

The ***governance system*** under which decisions are made, and the degree to which power over a given decision is shared among ***actors*** or across different ***sectors***, contributes significantly to the types of decision support, scenarios and models that are useful. For example, 'top-down', 'single-actor' decision problems may be amenable to the application of economic optimisation approaches, while more '***participatory***', 'multi-actor' decision processes may be better supported by deliberative approaches such as multi-criteria mapping (Stirling and Mayer, 1999; De Marchi and Ravetz, 2001). Other aspects of governance that determine how a decision will play out include the ***history*** and ***legitimacy*** of the governing institutions.

The ***time horizon*** for which a decision is expected to hold and the ***frequency*** of decision making about a particular issue have a large influence on the sorts of scenario, modelling and decision-support approaches that may apply. Sequential decision processes provide the opportunity to value the role of learning and to establish formal programmes of 'continuous improvement', often invoking ideas embodied in adaptive management (Walters, 1986). However, with this opportunity comes complexity. Many reasons have been proposed for the conspicuous lack of working examples of adaptive management in broad-scale, multi-objective decision problems, including a reluctance to set measurable management objectives, a reluctance to invest in long-term monitoring of management outcomes, and a reluctance to formalise assumptions about cause-effect pathways as testable hypotheses or models (Walters, 2007; Wintle and Lindenmayer, 2008; Westgate et al., 2013).

Most environmental decisions are characterised by multiple competing views about what constitutes a good outcome (Keeney, 2007). This arises because different stakeholders hold different ***objectives***, which imply different criteria by which outcomes will be measured. Decisions that involve multiple objectives will tend to be more difficult to make than decisions for which there are few objectives. One of the reasons why people hold different objectives for a particular decision problem is that they share different ***values***. Decision problems characterised by multiple zvalues tend to be much trickier to resolve than when values are shared. A common challenge to decision making in many parts of the world arises because not all ***stakeholders*** share the same ***knowledge system***. Very few analytical decision-support approaches, including scenario development and modelling approaches, are easily applied across multiple knowledge systems, although more deliberative, participatory processes tend to be favoured in such circumstances.

Differences in capacities and power determine the effectiveness of ***stakeholder*** representation and the acceptability of decision outcomes. Large and wealthy organisations, including companies and national governments, may have greater resources and better access to information than other stakeholders, leading to a greater influence over the decision process. Assessing the impacts of policies, plans and management options on livelihoods may require culturally-specific, local-level understanding to properly evaluate costs and benefits to all stakeholders (Nordström et al., 2010; Rowland et al., 2014; Runge et al., 2011b). Cultural norms, values, practices, ideologies and customs shape people's understanding of their needs, rights, roles and possibilities, and hence influence their actions, including engagement in policy design and implementation (Borrini-Feyerabend et al., 2004). All stakeholders use their beliefs as the basis for determining the range of options they will consider and the criteria by which they will measure outcomes. The importance of taking into account multiple belief systems during policy formulation is being increasingly recognised, especially in areas where indigenous people have consolidated their property and representation rights (TEBTEBBA, 2010; UN, 2008; Runge et al., 2011b).

Uncertainty takes many forms (Regan et al., 2002) and impacts on environmental decisions in a variety of ways (Section 2.3.3, Ludwig et al., (2001)). Uncertainty can arise due to a lack of ***information***, either in the form of traditional and ***scientific knowledge***, ***data*** and/or ***capacity***, or simply due to high levels of environmental and ecological ***stochasticity***, as well as a variety of other sources (Section 2.3.3, Regan et al. (2002)). The degree and type of uncertainty inherent in a particular decision problem determines the sorts of analytical and decision-support approaches that can be applied (Peterson et al., 2003; Regan et al., 2005) and partly motivates the need for scenarios and models. The role and implications of uncertainty in decision making, scenarios and modelling are dealt with in Section 2.3.3.

A high level of decision complexity provides a strong motivation to utilise decision-support approaches because the complexity of many decisions exceeds the processing capacity of the human brain. Aside from the social, cultural and governance complexities already mentioned, ***ecological*** complexities such as the ***heterogeneity*** of ecosystems, the ***diversity*** of species involved and the degree to which decision have to address ***cross-landscape*** flows and connections make for more or less tractable decision contexts. Some 'local' decisions take place within a particular ecosystem or geographical domain that can be considered – for the purposes of the decision process – discrete and sufficiently buffered from the ecological processes playing out in other systems, so as to simplify the characterisation of biodiversity and ecosystem service values and dynamics. However, many land-use planning and policy processes play out over multiple ecosystems that are connected by complex flows of biotic and abiotic resources, and that are subject to multiple types of ecological and social dynamics that may play out over multiple temporal scales. For example, some integrated catchment management strategies must consider simultaneously terrestrial, river, estuarine and near-shore ocean ecosystems, each with unique economic drivers and pressures such as agriculture, aquaculture and fishing (e.g. Brodie et al., 2012).

***Spatial and temporal scale***, including the spatial and temporal grain and extent relevant to a particular problem, drive the level of modelling, scenario and decision-support sophistication required to support decisions. Biodiversity and ecosystem services have specific spatial and temporal distributions that overlap with human management units or jurisdictions in complex ways. Similarly, stakeholders have rights, obligations and interests at a variety of spatial scales, making ***cross-scale dynamics*** an important part of the decision context. Global responses to ecosystem problems are warranted when those problems potentially affect all people and ecosystems. Multilateral, regional and bilateral agreements require consensus by a group of nations but implementation often requires action within national boundaries. National policies exist independently of agreements with other nations, highlighting the problem of policies and plans that conflict across scales. The scale at which human and biotic processes operate influences the sorts of decision approaches, scenarios and models relevant to a particular decision. The spatial scale partly determines who will be represented in a decision problem and whose interests are considered.

**Table 2.1:** Attributes that define a decision context and how they vary.

| | Decision context attributes | Easier to decide | More difficult to decide | Decision strategies for easier decisions | Decision strategies for more difficult decisions |
|---|---|---|---|---|---|
| **Governance** | *Actors* | Single/executive | Multiple/negotiated | Optimisation, benefit cost analysis | Multi-criteria decision analysis (mcda) |
| | *History* | History of governance | Novel governance | Learning | Assessment |
| | *Legitimacy* | Accepted | Contested | Executive | Conflict resolution |
| | *Sectors* | Single | Multiple | Executive | Negotiation and bridging |
| | *Participation* | Consultation | Decision | Communication | Participatory decision making |
| **Decision** | *Decision time horizon* | Short term (months) | Longer term (decades) | Optimisation | Adaptive management |
| | *Decision frequency* | One-off | Repeated | Assessment | Monitoring and learning |
| | *Objectives* | Single | Multiple | Optimisation | Mcda, analytic hierarchy process (ahp) |
| **Stakeholders** | *Values* | Homogenous | Diverse | Assumed | Deliberation and negotiated |
| | *Knowledge system* | Homogenous | Diverse | Single process | Bridge multiple processes |
| **Information** | *Scientific knowledge* | High | Low | Optimisation, benefit cost analysis | Adaptive management |
| | *Data availability* | High | Low | Optimisation, benefit cost analysis | Delphi, robustness, deliberation |
| | *Scientific capacity* | High | Low | Optimisation, benefit cost analysis | Deliberation, participation |
| **Ecology** | *Heterogeneity* | Single ecosystem | Multiple ecosystems | | |
| | *Diversity* | Single species | Multi species | | |
| | *Flows across landscape* | Weak connections | Strong connections | | |
| | *Stochasticity* | Low and predictable | High and unpredictable | | |
| **Scale** | *Cross-scale dynamics* | Weak external influence | Strong external influence | | |
| | *Temporal extent* | Short-term | Long-term | | |
| | *Temporal grain* | Seconds | Millennia | | |
| | *Spatial extent* | Local | Global | | |
| | *Spatial grain* | Metres/seconds | Kilometres/degrees | | |

## 2.3 Overview of agenda-setting and decision-support approaches

Many methods, approaches and tools exist to support activities in each phase of the policy cycle. A broad distinction is drawn between tools that support policy agenda setting (Section 2.3.1) and tools that support actual decisions in the policy design, implementation and review phases of the policy cycle (Section 2.3.2).

While the scenarios and models used in these two activities may be similar or identical, there are important differences in the way they are used that arise due to differences in the agenda-setting versus policy-design, implementation and review contexts. A non-exhaustive overview of the main families of agenda-setting and decision-support approaches is provided. Families of decision-support approaches are described in rough order of complexity, ranging from relatively generic tools to more highly integrated frameworks (Section 2.3.2). Case studies of the application of several approaches are provided as boxed essays. A table documenting how each approach fits within the decision-context typology is provided in Section 2.4. A database of case studies documenting applications of each decision-support approach according to decision-context variables will be provided to the IPBES scenarios and models expert group (Deliverable 3c) and the IPBES deliverable on policy and decision tools (Deliverable 4c).

### 2.3.1 Policy agenda setting

Agenda setting is one of four phases in the policy cycle (Figure 2.1) that motivates and sets the direction for policy design and implementation. Scenarios and models often play a role in agenda setting.

The first GBO (sCBD, 2001) presented information from national reports and a global evaluation of biodiversity trends (WCMC, 1992). These analyses were later augmented with exploratory scenario analysis in the second GBO – the *Crossroads of Life on Earth* study (sCBD and PBL, 2007). This study used GLOBIO as a modelling framework to assess the impact of environmental drivers on biodiversity and explore policy options in the form of intervention scenarios to reduce biodiversity loss and achieve the 2010 targets for biodiversity. The third GBO (sCBD, 2010) also presented biodiversity scenarios and tipping points contained in a study incorporating the results of the Millennium Ecosystem Assessment (MA), the GBO 2 and the Global Environment Outlook (GEO) 4, as well as the Mini Climate Assessment Model (MiniCAM) (Leadley et al., 2010). The fourth GBO provides a mid-term assessment of progress towards the implementation of the *Strategic Plan for Biodiversity* and achievement of the Aichi biodiversity targets (Alkemade et al., 2009; Leadley et al., 2014). These assessments have all contributed significantly to the current policy agenda pertaining to biodiversity and ecosystem services at multiple spatial scales across multiple jurisdictions (Figure 2.2).

Global agendas play out at regional and national scales in many ways. Referring directly to the Convention on Biological Diversity (CBD), the National Performance Assessment and Sub-regional Strategic Environment Framework for the Greater Mekong Sub-region (GMS; ADB, 2010) was developed to guide the GMS Core Environment Programme, through which the GMS governments create a vision and framework for long-term investment in environmental governance, institution building, environmental protection in the main development sectors, and biodiversity conservation. In this process, the GLOBIO3 model underpinned the assessment of different policy options to reach biodiversity targets in the region (Figure 2.2).

At a regional level, the European Commission developed the European Union (EU) Biodiversity

Strategy (EC, 2011), which was informed by an assessment of the 2010 biodiversity targets (EEA, 2009). These activities represent policy formulation and evaluation, following from the agenda set by CBD and MA (Figure 2.2). International fisheries policy in the same region has been influenced by models and scenarios at the same scale (Box 2.1). At a local scale, 20-year Forest Agreements were signed between the Australian government (responsible for implementing the Environment Protection and Biodiversity Conservation Act (1999) and export licencing) and the New South Wales state government (NSW, responsible for land management) that set out new forest conservation reserves and approved ecologically-sustainable forest management systems in four regions across the state. The negotiation of these agreements was based in part on C-Plan (Pressey et al., 2009), a participatory land-use planning decision support tool that utilises species distribution models (Ferrier et al., 2002) and forest growth and yield models (Vanclay, 1994) to identify trade-offs between forestry production and species conservation objectives.

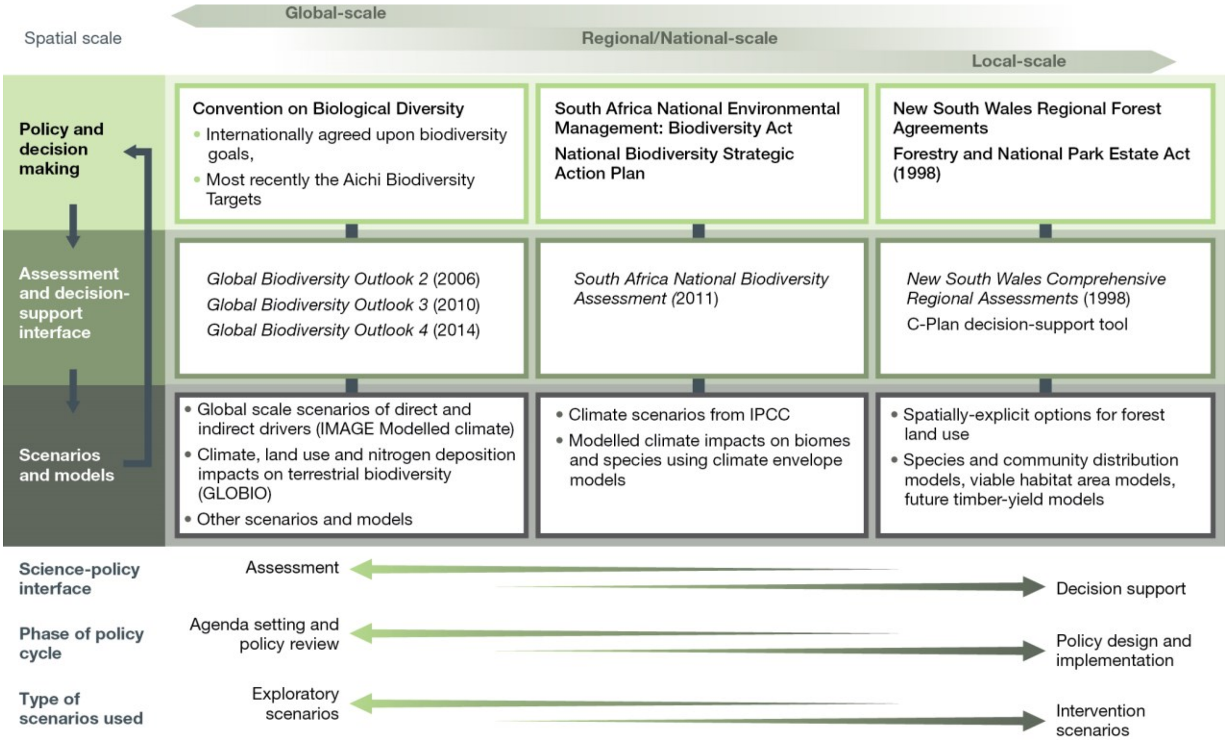


**Figure 2.2:** Commonly observed relationships between spatial scale, phase in the policy cycle and model or scenario type using Aichi biodiversity targets and subordinate activities as an example. At the global scale, CBD and Aichi biodiversity targets were partly informed by assessments, models and scenarios at that scale. Numerous subordinate processes at regional, national and local scales draw on the CBD and Aichi biodiversity targets to motivate policies, plans and actions. Lower-level activities also draw on combinations of global (and finer) scale analyses, in concert with decision-support protocols (Section 2.3.2) to design and implement policies. Both top-down and bottom-up modelling and scenario analysis approaches can support decision making at regional, national and local scales (Chapter 6). For example, the South Africa National Biodiversity Strategy and Action Plan (Deat, 2005) guides policy design and implementation at finer scales and was informed by the National Biodiversity Assessment (Driver et al., 2012), which used bioclimatic models to incorporate climate resilience into species and ecosystem planning. In New South Wales, Australia, correlative species distribution models and forest growth models were combined using participatory decision-support software (C-Plan) to generate spatial land-use options for forestry and conservation objectives in four regions during the comprehensive regional assessment that preceded the regional forest agreements. It is acknowledged that, while there is no one-to-one correspondence between spatial scale and policy cycle phase or scenario type, this scheme does provide some insight into commonly observed hierarchies of policy, planning and action and some of the tools that are used at different levels in the hierarchy. For example, there could be a role for formal

decision-support protocols such as the Delphi or structured decision-making approaches (Section 2.3.2) in setting Aichi biodiversity targets, but there is no documented evidence of this occurring in that process.

**Box 2.1:** Models and scenarios for policy agenda setting at a regional scale: European international fisheries policy

The European marine policy frameworks have adopted ecosystem-based management, which requires indicators that describe pressures affecting the ecosystem, the state of the ecosystem, and the response of managers (Jennings, 2005); Figure Box 2.1). This adoption of ecosystem-based management is due to a shift in research effort from single species to ecosystem-based concerns, reflecting a growing recognition that an ecosystem approach may help to underpin improved management (Jennings, 2004). Numerous published models describing the complexity of marine ecosystems (Baird et al., 1991; Baird and Milne, 1981; Baird and Ulanowicz, 1989; Piroddi et al., 2015) underpin indicators that drive the Marine Strategy Framework Directive (MSFD 2008/56/EC, EU, 2008) that arose out of the Common Fisheries Policy (1982) (European Parliament, 2009). The Marine Strategy Framework Directive requires that EU Member States achieve 'Good Environmental Status' under 11 descriptors of the marine environment by 2020. Of these 11, descriptor 4 (D4) addresses marine food webs: 'All elements of the marine food webs, to the extent that they are known, occur at normal abundance and diversity and levels capable of ensuring the long-term abundance of the species and the retention of their full reproductive capacity'. The D4 indicator stipulated in the Commission Decision (EC, 2010; Rogers et al., 2010) addresses three criteria related to food web structure and energy transfer. Descriptor 1 on biodiversity also relates to species distribution ranges, habitat extent, habitat condition and ecosystem structure. Many of these measures are dependent on habitat and ecosystem models, as few are directly measurable at broad scales in the marine environment.

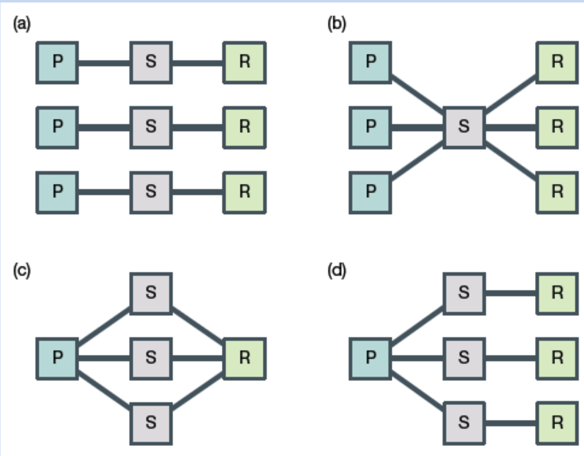


**Figure Box 2.1:** Possible relationships between pressure (P), state of the ecosystem (S) and response to a management action (R). Figures (b)-(d) illustrate that indicators of P, S and R are rarely expected to map one-on-one as in (a) (Modified from Jennings (2005). *Indicators to support an ecosystem approach to fisheries.* Copyright © 2005 by John Wiley Sons, Inc. Reprinted by permission of John Wiley & Sons, Inc).

## 2.3.2 Families of decision-support tools

A myriad of methods and approaches exist to support the policy design, implementation and review phases of the policy cycle. Methods and approaches exist within a multi-dimensional '***decision context***' (Figure 2.3), defined in part by ***decision-context attributes*** (Table 2.1).

Tools and approaches range from technical tools within a very specific domain of application such as mathematical optimisation approaches, through to broad frameworks such as 'structured decision making' (Gregory et al., 2012) and adaptive management (Walters and Holling, 1990) that provide flexibility for dealing with most challenges confronting environmental policymakers and managers.

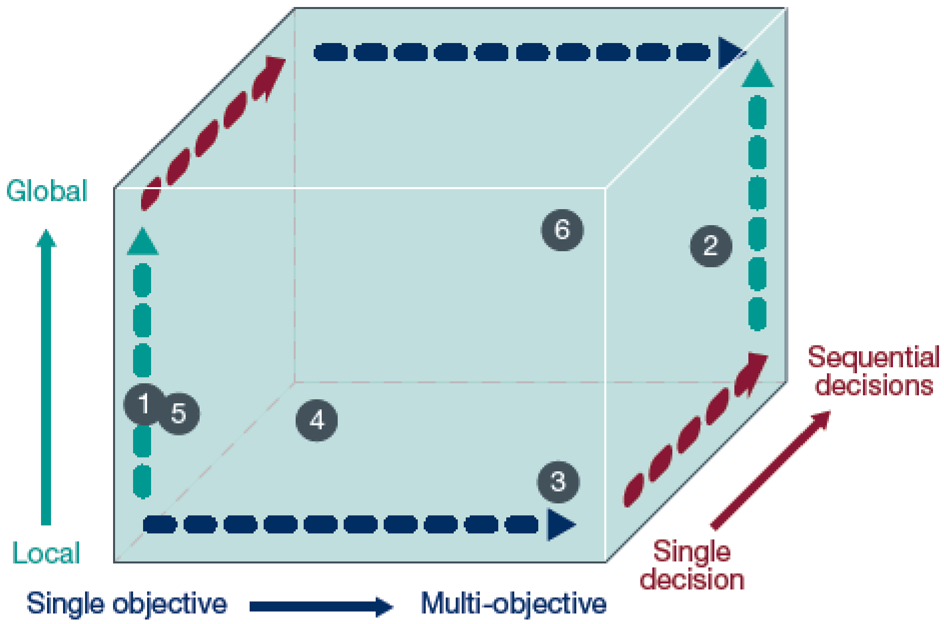


**Figure 2.3:** Three dimensions of decision context. Dashed arrows indicate increasing complexity from a single (one-off) decision made by a single group with a single objective at a local scale, to a sequential decision made by a group of decision makers with multiple (usually competing) objectives at regional/global scales. Numbered circles indicate individual applications of a given decision-support method, undertaken in different parts of the decision space. For example, circle 1 represents a study (Joseph et al., 2008) in which a single organisation (NZ DoC) used a single objective criterion (maximise increase in species persistence/$) at the national level. Circle 5 identifies a conservation planning exercise, undertaken by the Malagasy governments, with the single objective of identifying the areas of Madagascar that would most efficiently increase the representativeness of the Madagascar reserve system (Kremen et al., 2008). There was no explicit consideration of sequentially increasing the reserve system or the multiple competing social or cultural objectives in the structured part of the reserve design process, though these considerations would likely have played out in the less structured political process. In contrast, study 2 reports on a decision process in which multiple cultural groups with multiple (incommensurable) objectives participated in a decision about the control of non-native fish species in the Glen Canyon Dam in southern USA (Runge et al., 2011b). Study 2 was described as a 'structured decision-making' exercise (Section 2.3.1.4; Gregory et al., 2012), supported by MCDA with swing weighting to help identify dominated options. Study 3 provides an example of a once-off, multi-objective decision problem at a local scale (Box 2.2; Mustajoki et al. 2004), while circle 6 could represent a global, multi-objective, one-off policy decision, such as the establishment of the UN Sustainable Development Goals (https://sustainabledevelopment.un.org). No value judgment is implied by this figure about where in decision space is the best place to be; the point to note is that different decision-support approaches suit different parts of the space.

The following sections review a sample of decision-support methods that occupy different parts of the decision-context space (Figure 2.3). The case studies presented were chosen from 91 examples found in grey and peer-reviewed literature during a non-exhaustive search by the authors. Consequently, this is not an exhaustive inventory of methods, nor does it cover all parts of the decision-context space. The aim is to provide an entree to a range of commonly used decision-support methods, frameworks and approaches and to discuss the role of scenarios and models in each.

### 2.3.2.1 Multi-objective approaches to analyse trade-offs

Most decision making involves, either implicitly or explicitly, the analysis of risk. Risk is generally considered to be the product of likelihood and consequence (Burgman, 2005), which is essentially an estimate of expected utility (Savage, 1954). While consideration of adverse consequences alone will often suggest the desirability of risk avoidance or mitigation measures, conditioning estimates of consequence with assessment of likelihood may lead to the conclusion that risk avoidance or mitigation are not warranted (because likelihoods are sufficiently low). If estimates of likelihood and consequence are accurate, then decisions based on risk should lead to the more efficient allocation of resources than considering only consequences (Arrow and Lind, 1970). Risk assessment approaches are used widely in environmental decision making (Burgman, 2005). Risk analysis forms the basis of Environmental Impact Assessments (EIA; Section 2.3.2.3) and many of the integrative decision-support approaches reviewed in this chapter. Risk analysis is needed anytime there is uncertainty that cannot be reduced, that is, when decisions have to be made in the face of risk.

The real-world challenges of decision making are seldom simple, with high decision complexity being the norm in most decision contexts (Table 2.1). Consequences are seldom restricted to impacts that can naturally or readily be described by a single criterion (e.g. monetary). Multiple values imply multiple objectives each requiring estimates of consequence.

Uncertainty about consequences and likelihoods brings into play complex risk preferences that must be considered. Most decisions involve alternatives and cause-and-effect predictions of expected consequence, providing a natural role for scenarios (to characterise alternatives) and models (to predict consequences). When predictions are made over multiple objectives, an additional element is required to resolve the decision problem: the articulation of preferences or trade-offs reflecting the relative importance of the different objectives (Howard, 2012). Most environmental policy, planning and management decisions involve trade-offs (Keeney, 2007).

Single-attribute risk management tools do not directly treat trade-offs among competing objectives. A subset of these tools may be helpful in prompting exploration of cause-and-effect relationships using models during the process of estimating expected consequences for individual options or objectives, but on their own they will generally be inadequate for making most real-world decisions that tend to involve trade-offs.

***Consequence tables*** are the first of the multi-objective decision-support tools described here to deal explicitly with trade-offs. There are three core elements to any multi-objective decision problem; alternatives, expected consequences and trade-offs. These elements are compactly reported in a consequence table. An example is shown below (Table 2.2), where alternatives comprise six hypothetical candidate options for reducing impacts on a near-shore reef system resulting from nutrient outflow from an agricultural catchment. The table can be populated with qualitative or quantitative estimates of expected consequence. Experts and non-expert stakeholders alike are notoriously deficient in their capacity to make internally-consistent probabilistic judgments (Hastie and Dawes, 2010). Modelling tools that assist in the coherent treatment of probabilities include fault tree analysis, event tree analysis, Markov analysis, Monte Carlo simulation and Bayes nets. For example, Jellinek et al. (2014 developed a Bayes net to predict the relative improvement in vegetation condition resulting from a range of woodland management intervention scenarios such as reducing stock grazing and undertaking vegetation restoration.

**Table 2.2:** The example below uses coarse verbal (negative) impact descriptors typically seen in a qualitative risk matrix approach. Trade-offs involve consideration of the performance of each alternative against each objective. The top row represents six hypothetical candidate management options, and the first column gives each objective (criteria) against which expected consequences are assessed.

| Example objectives | Do nothing | A1 | A2 | A3 | A4 | A5 | A6 |
|---|---|---|---|---|---|---|---|
| Biodiversity – fish | High | High | High | Medium | Medium | Low | Low |
| Biodiversity – coral | Extreme | High | High | Medium | Medium | Low | Low |
| Economic cost | Low | Low | Medium | Medium | Medium | High | Extreme |
| Costs to implement | Low | Medium | Low | Low | High | Extreme | High |
| Recreational fishing | High | Medium | Medium | Medium | Medium | Medium | Low |
| Tourism | High | High | High | Medium | Medium | Low | Low |

The preparation of a consequence table itself offers substantial insulation against the pitfalls of unaided decision making. However, unless the decision problem can be meaningfully simplified to two or three objectives and two or three alternatives, the cognitive and emotional demands on decision makers and stakeholders can lead to poor outcomes such as environmental impacts that could have been avoided at little cost to development. In many instances, a consequence table can be simplified through the identification of the *strictly* non-dominated set of alternatives (options for which no single alternative is better according to all criteria) and redundant objectives. An alternative is strictly dominated if, in comparison with any other single alternative, it performs worse on at least one objective and no better on any other objective. Driscoll et al. (2015) identified non-dominated sets of management strategies in a trade-off between asset protection, the provision of three ecosystem services (water provision, carbon sequestration and atmospheric pollutants) and the conservation of four species in the context of wildfire management. Identifying a set of non-dominated options that represents a range of trade-offs between two or more criteria is also known as ***Pareto analysis*** (Chankong and Haimes, 1983). The set of options identified as the non-dominated set for a range of trade-offs between two criteria comprise the ***Pareto frontier*** (Chankong and Haimes, 1983; Driscoll et al., 2015).

If all expected consequences can be assigned a monetary value, then ***cost-benefit analysis*** (also known as ***benefit-cost analysis***) may be applicable. Selection of the option with the highest benefit-cost ratio has a strong basis in public policy and welfare economics. However, the monetisation of non-market values is difficult and some implementations of cost-benefit analysis avoid monetisation by seeking an alternative common currency.

Many applications of cost-benefit analysis rely on revealed preferences data (what people are prepared to pay). Where revealed preferences are deemed inadequate or absent, techniques for stated preferences are available (Bennett and Blamey, 2001), but the time and resources required to apply these methods are substantial. In many cases, stakeholders are unlikely to feel comfortable with the monetisation of all objectives, especially those dealing with social and environmental outcomes (Jax et al., 2013. Cost-effectiveness analysis is a variant of cost-benefit analysis that accommodates the non-monetary comparison of options. For example, Joseph et al., (2008) utilised cost-effectiveness analysis to prioritise threatened species conservation projects in New Zealand on the basis of extinction risk reduction achieved per dollar, weighted by phylogenetic uniqueness.

Maguire (2004) cites two interacting flaws commonly encountered in risk-based decision support: a) incoherent treatment of the essential connections between social values and the scientific knowledge necessary to predict the likely impacts of management actions, and b) reliance on expert judgment about risk framed in qualitative and value-laden terms, inadvertently mixing the expert's judgment about what is likely to happen with personal or political preferences.

***Multi Criteria Decision Analysis*** (MCDA) (Keeney and Raiffa, 1976) is a way of analysing trade-offs between decision options according to multiple objectives (criteria). The family of techniques under the banner of MCDA seeks to avoid the flaws in risk-based decision support identified by Maguire (2004) by explicitly separating the tasks of causal judgment (what might happen and why) and articulating value judgments or trade-offs (how one values particular outcomes: Ananda and Herath, 2009) (see example in Box 2.2).

***Multi-Attribute Value Theory*** (MAVT) is a foundational idea in MCDA. Applications of MAVT seek to describe a decision maker's value function over two or more objectives and associated criteria:

$$v(x_1,\ldots,x_n) = \sum_{i=1}^{n} w_i v_i(x_i),$$

where $w_i$ are the weights and $v_i$ are value functions for any single attribute. Weighting of the individual value functions can be done formally by the method of indifferences, akin to the underpinnings of stated preference techniques used in the evaluation of non-market impacts in benefit-cost analysis (Bennett and Blamey, 2001). There are many shortcut methods for eliciting weights (Hajkowicz et al., 2000). Of these, the swing weight method has been shown to be one of the more effective, both in terms of its efficiency and its insulation against abuse (Fischer, 1995). Whatever method is used in their elicitation, the interpretation of the weights is critical. Methods that do not explicitly deal with indifferences are prey to abuse, as users are inclined to specify weights that reflect the relative importance of the attributes, irrespective of the units or the range of consequences relevant to the decision context. However, the weights have units because the underlying attribute scales have units. Changing the units or range of an attribute *must* lead to a change in the weights. For the *additive* value model to be valid, the attributes need to be *mutually preferentially independent*. In practice, the assumption of preferential independence is reasonable if the set of objectives is complete, non-redundant, concise, specific and understandable (Keeney, 2007). Where objectives satisfy these properties there is a strong case for the use of simple weighted summation. While the analyst needs to be careful to ensure preferential independence, the mechanics of MAVT are straightforward, with arithmetic operations simple and easy to implement in a spreadsheet. Because MAVT is based on point estimates of consequence, it is strictly speaking only applicable where there is no uncertainty in the estimation of consequences or where decision makers and stakeholders can be assumed to be risk-neutral, such that value judgments are restricted to a consideration of mean expectations rather than the full set of possible consequences encompassed by worst-case and best-case scenarios. Comprehensive descriptions of MAVT are provided by Bedford and Cooke (2001) and Keeney (2007).

***Analytic Hierarchy Process*** (AHP) is an MCDA application commonly encountered in the natural resource management literature (Mendoza and Martins, 2006). It is essentially a variant of MAVT designed to minimise the elicitation burden on experts and decision makers. Most applications employ the same additive value model described above for MAVT. Using a nine-point preference scale and matrix computations to translate ordinal judgments into cardinal judgments, a) marginal value functions, and b) weights are derived through pairwise comparisons of alternatives and objectives respectively (Saaty, 1980). A variety of software packages are available, although for simple problems the calculations can be done in a spreadsheet.

AHP's strength in minimising the elicitation burden is also its weakness, as it is possible to obtain marginal value functions without any explicit estimation of consequences. For decision problems involving self-evident cause-and effect relationships this may be acceptable. However, this may fall down when consequences of alternative options involve difficult probabilistic judgments that are likely to be logically challenging (Hastie and Dawes, 2010).

AHP has also been criticised on theoretical grounds because it allows rank reversal upon introduction of a new alternative (Belton and Gear, 1983). The modified AHP is free of this problem as it uses standard MAVT techniques to obtain marginal value functions and limits the use of pairwise comparisons to the derivation of weights. (Moffett and Sarkar, 2006) advocates use of the modified AHP because of the relative ease of obtaining weights. However, like direct weighting, weights obtained through pairwise comparisons via the modified AHP result in the poor capture of stakeholder preferences. In general, respondents tend to assign weights according to the perceived importance of objectives, irrespective of the consequences associated with the specific alternatives being considered, which is considered inadequate under conventional decision theory (Steele et al., 2009).

***Outranking*** techniques stem from the French school of MCDA, which places less emphasis on normative understanding (assuming an ideal decision maker who is rational, fully informed and able to compute accurately) of how decisions *should* be made based on axioms of rationality (von Neumann and Morgenstern, 1944) and greater emphasis on behavioural models of decision making (Roy, 1973). Outranking techniques typically involve the sequential elimination of alternatives (Chankong and Haimes, 2008). Weights are assigned to each objective according to their perceived importance, without consideration of the range of consequences associated with alternatives. For each pair of alternatives, a concordance index and a discordance index are constructed. The concordance index coarsely characterises the strength of the argument that one alternative is better than another based on the weighted sum of objectives for which it dominates the other. The discordance index reports the strength of the argument against eliminating the (weakly) dominated alternative. Decision makers work through a consequence table iteratively, adjusting critical thresholds for concordance and discordance until a satisfactory choice is made.

There are numerous techniques and software packages that fall under the banner of outranking (e.g. ELECTRE, PROMETHEE, GAIA; see Figueira et al. (2005 for details). The techniques vary according to how expected consequences are characterised. If a consequence table is populated using qualitative ordinal descriptors of impact (e.g. Table 2.2), ELECTRE can informally support stakeholders process trade-offs and difficult decisions involving more than a handful of objectives and alternatives. While other outranking techniques can be used where consequence estimates are quantitative or semi-quantitative, there is little argument for doing so, because in these circumstances MAVT offers a much firmer normative basis for decision making.

The formal description of ***Multi-Attribute Utility Theory*** (MAUT) developed by von Neumann and Morgenstern, 1944 remains a high point in the theory of MCDA. It is also a wholly impractical approach to typical multi-objective, multi-stakeholder problems. Many of the developments and refinements of MCDA that have taken place since the 1950s are essentially pragmatic shortcuts for MAUT. MAUT can be used when a consequence table is populated by statistical distributions describing probabilistic uncertainty in the performance of each alternative against each objective. In this way, MAUT provides a link between MAVT and risk analysis, allowing both the multiple-objective and the risk (utility) tools to be brought to bear on a problem. Given that many real problems contain these features, this can be considered a good thing. However, the circumstances in which this can be achieved are rare indeed, especially in natural resource management. Aside from difficulties in obtaining detailed probabilistic causal judgments, there are distinctly onerous demands on decision makers and stakeholders in the elicitation of trade-offs under MAUT. Populating a consequence table with probabilistic outcomes clearly defines a strong role for scenarios and models. In practice, only the most committed and indefatigable participants in group decision-making settings are capable of formally addressing trade-offs using MAUT, highlighting the importance of technical modelling and scenario analysis support for the successful implementation of such approaches.

**Box 2.2:** Multi-criteria decision analysis (MCDA) case study – the use of a web-based MCDA system in participatory environmental decision making in Finland

Mustajoki et al. (2004) describes the use of MCDA in planning for multiple uses of the Paijanne Lake – Finland's second largest lake. The lake has been regulated since 1964, with the original objectives being to increase hydropower production and decrease agricultural flood damage. The lake has extensive recreational housing developments along its shore and there are tens of thousands of recreational users and fishermen on the lake. There has been growing public interest to reconsider the regulation policy to better take into account the increased recreational use and current high environmental awareness. Problems currently recognised on the lake include the low water levels during spring, changes in the littoral zone vegetation and the negative impacts of the regulation on the reproduction of fish stocks. An extensive multidisciplinary research project was carried out between 1995 and 1999 to re-evaluate the regulation policy of the lake. The aims of the project were to assess the ecological, economic and social impacts of the regulation. Stakeholder opinions were sought about the current regulation and its development, a comparison of new regulation policy options, and recommendations to diminish the harmful impacts of the regulation (as, for example, in Figure Box 2.2). An open and participatory planning process was considered necessary to gain public support for the project and to find consensus on a new regulation strategy. A steering group consisting of 18 representatives of different stakeholders was set up by the Ministry of Agriculture and Forestry, the permit holder of the regulation license. Additionally, four working groups were established to improve communication between the water resource authorities, local stakeholders, regulation experts and researchers. To inform the public, a local press conference was arranged after almost every steering group meeting. In a survey of participants, 80% agreed that 'the recommendations for the regulation were able to combine the different and conflicting interests of both the people living on the lake and the downstream water system'.

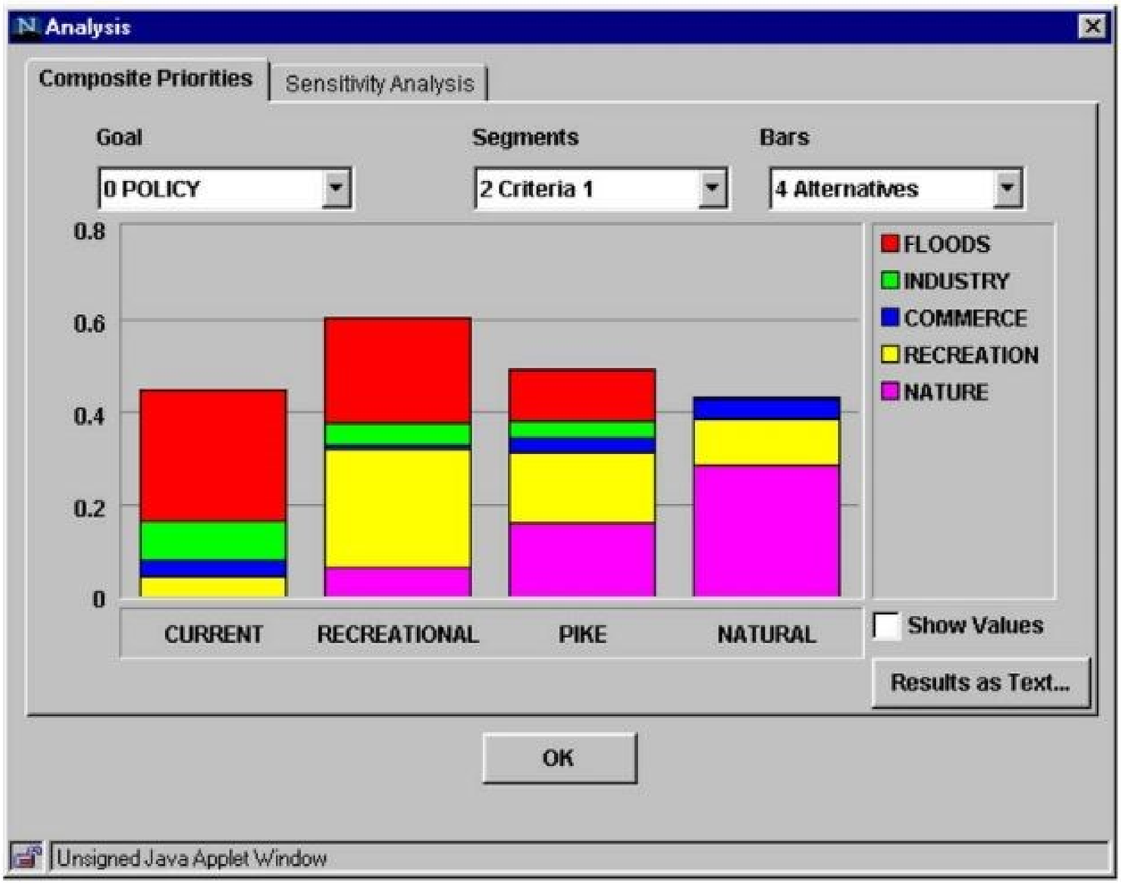


**Figure Box 2.2:** An example of the overall values of alternative management scenarios as determined by stakeholders from various sectors (Reprinted from Environmental Modelling & Software, 19/6, Mustajoki et al. (2004), Participatory multicriteria decision analysis with Web-HIPRE: a case of lake regulation policy, 537-547, Copyright 2004, with permission from Elsevier).

### 2.3.2.2. Optimisation approaches

There are potentially thousands of alternative options in most real-world planning and management decision problems. Various mathematical programming techniques from the field of operations research are available to help identify better (or best) candidates from a large set (Chankong and Haimes, 2008). Optimisation approaches can be viewed as providing the analytical machinery to assist in the generation and analysis of 'target-seeking' or 'backcasting' scenarios (Chapter 3).

Optimisation problems are framed with a decision set, an objective and constraints. Depending on the characteristics of the problem and the relationship between the actions and the expected consequences (e.g. linear, convex, smooth or non-smooth, dynamic or non-dynamic, deterministic or governed by uncertainty), there are various classes of resolution method (Chankong and Haimes, 2008). A small sample is reviewed here. Two such classes of resolution method include ***linear programming*** and ***stochastic dynamic programming***, which employ algorithms designed to optimise an objective function under specified constraints (Chankong and Haimes, 2008). In linear programming, a linear (or near-linear) relationship between actions and expected consequences is required. This may be inappropriate in many ecosystems, where outcomes for objectives are dynamic and non-linear in relation to actions or sets of actions. Both linear programming and stochastic dynamic programming are single-objective optimisation approaches. Multi-objective problems can be partly accommodated with the use of extra constraints. With a detailed understanding of cause-and-effect, stochastic dynamic programming can accommodate non-linear, dynamic outcomes associated with stochastic risk (e.g. risks associated with wildfires) superimposed on the deterministic influence of management actions (e.g. fuel reduction burning in high fire risk places).

Stochastic dynamic programming recognises that what might be considered a desirable action depends on the state of the system (Minas et al., 2012; Richards et al., 1999). For example, a low fire risk and a strong social preference for minimal management that impacts on natural values may imply lower preference for risk reduction (planned) burning compared to circumstances where fire risks are high (lots of woody debris) and public concern about active forest management is low. The capacity to capture greater realism in Stochastic Dynamic Programming is attractive, but computational overheads, the curse of dimensionality (inability to deal with very large numbers of possible states) and the requirement for sophisticated causal understanding mean that most applications are substantially simplified. ***Goal programming*** requires specification of a performance aspiration for each objective, and the underlying algorithm searches among the candidates for the alternative with the minimum multi-dimensional distance to the goal set (Chankong and Haimes, 2008). A single decision maker can use the method profitably however, in a multi-stakeholder setting, goal programming is open to abuse because stakeholders will tend to manipulate outcomes through the articulation of insincere positions on what might be considered an appropriate goal for each objective.

***Integer linear programming*** is able to address many non-linear optimisation problems by using linearisation techniques and commercially available integer linear programming solvers. Substantial progress has been made in the field of non-linear mathematical programming with continuous or integer variables. Numerous optimisation problems can be formulated within this framework and articles published in the conservation and biodiversity protection are based on these techniques (Billionnet, 2013).

***Heuristics*** are often used for decision optimisation problems that cannot, for a variety of reasons including high complexity or size, yield exact optimal solutions. Examples of commonly used heuristics include ***simulated annealing***, ***Tabu search*** and ***genetic algorithms*** (Dréo et al., 2006). Graph theory is also a powerful tool for modelling and solving optimisation problems (Krichen and Chaouachi, 2014). Some commonly-used spatial conservation prioritisation approaches such as ***Zonation*** (Moilanen et al., 2005) and ***Marxan*** (Possingham et al., 2000) utilise heuristics.

#### 2.3.2.3 Integrative approaches

This last family includes a large number of frameworks, approaches and methods, few of which can be described here. Multiple variants exist for every approach described, often with similar structures and underpinnings, but with different names arising from their application in different sectors (e.g. forestry, fisheries, transport) or regions. A brief overview of integrative approaches is provided here.

***Scenario planning*** – scenarios, as defined in Chapters 1 and 3, are now routinely incorporated in a wide range of decision-support approaches, including integrative approaches such as *management strategy evaluation* or *structured decision making* (Little et al., 2011; Section 2.3.1.4). Scenario analysis provides a framework in which to explore, characterise and organise uncertainties across spatial scales (Biggs et al., 2007).

Early developments in scenario analysis led to a particular decision-support approach known as ***scenario planning*** (Schoemaker, 1995). Börjeson (2006) refers to ***scenario planning*** as a tool for exploring possible, probable and/or preferable futures. Identifying strategies or options that are robust to a range of possible scenarios is also key in scenario planning (Peterson et al., 2003).

While scenarios are used in a wide range of agenda-setting activities and as part of integrated decision-support approaches, the relatively long history of scenario planning (Chermack, 2011, Schwartz, 1995) demands a specific mention here. Unlike forecasting, which aims to accurately predict future events, the focus of scenario planning is to explore possible futures that may arise under different conditions and what those different futures might mean for current decisions (Schoemaker, 1995). Assumptions about future events or trends are questioned, and uncertainties are made explicit (Bohensky et al., 2006). Scenario planning typically takes place in a workshop setting, in which participants explore current trends, drivers of change and key uncertainties, and how these factors might interact to influence the future (Schoemaker, 1993). To do so, they draw on both qualitative and quantitative information, including datasets (WCS Futures Group and BIO-ERA, 2007), spatially-explicit data (Santelmann et al., 2004) and expert/stakeholder judgment (Schoemaker, 1993). Based on this information, a set of plausible future scenarios is developed. Participants then consider a range of policy or response options and assess how robust those options are to the different scenarios developed (Box 2.3).

Shell Oil's navigation of the oil crisis of 1973 is an iconic example of the use of scenario planning, in which the company adjusted its business practices to buffer itself against the unlikely scenario of oil supply constraints (Peterson et al., 2003). In recent years, there have been many applications of scenario planning with a focus on biodiversity and ecosystem services on a landscape scale (Steinitz et al., 2003; Baker et al., 2004; Berger and Bolte, 2004; Hulse et al., 2004; Shearer, 2005; Walz et al., 2007; Patel et al., 2007; Santelmann et al., 2004). Others have combined scenario planning approaches with modelling approaches that incorporate human behaviour to better understand or characterise the effectiveness of policies or planning options (Happe et al., 2006; Bolte et al., 2006; Carmichael et al., 2004; Ittersum et al., 2008; Wei et al., 2009). Some studies (Liu et al., 2007; Meyer and Grabaum, 2008) have found a combination of optimisation and scenario analysis to be valuable in selecting land-use and management alternatives under uncertainty.

Strategy or option evaluation under scenario planning is commonly somewhat subjective. More formal decision-analysis methods can be used to support the evaluation of planning options under a range of scenarios (Goodwin and Wright, 2001), avoiding some of the pitfalls of subjective strategy evaluation. Schoemaker (1991) suggests that scenario planning should be used as a preliminary phase in the decision-making process, enabling the decision makers' ideas to be clarified, before moving to formal decision-analysis methods designed to support decision making under uncertainty (e.g. MAUT, Section 2.3.2.1), although reservations about this approach have been raised (Goodwin and Wright, 2001).

**Box 2.3:** Case study – scenario planning in the Hudson River Estuary watershed

In 2008, The Nature Conservancy worked with communities in the Hudson River Estuary watershed, USA, with the aim of preparing for the impacts of climate change (Aldrich et al., 2009; see also Cook et al., 2014a for further analysis). In a series of workshops over the course of 18 months, more than 160 stakeholders were consulted, including railroad executives, utility companies, the insurance industry, emergency and health groups, planners and conservation leaders. They identified and discussed important drivers (e.g. land-use trends, the political climate) and key uncertainties around those drivers (e.g. will there be strong 'top-down' political support for climate change adaptation?). By manipulating these uncertainties and trends, they created four plausible scenarios, which were described using suggestive titles (e.g. Stagflation Rules) and narrative details such as 'the early years of the scenario witness low to negative economic growth, falling real estate values and little new development in the region...'. Different elements of each scenario were specified; for example, the projections for the price of gas under the Procrastination Blues scenario were 'decline from $3.80 to $2.05 between 2008 and 2011, then rise rapidly back to $5.00/gal by 2016...'. The feasibility of different policies or response options (e.g. changing the requirements for new storm water permits) could then be evaluated, in terms of both the likelihood that they would be adopted in each scenario and how they would perform in each scenario. The 'top performing' options were those that scored relatively highly across the four scenarios (Table Box 2.3). This project provides a good example of the potential of scenario planning for evaluating intervention options. Focusing on the Hudson River Estuary watershed provided clear geographical scope and the drivers explored were well-defined and easily monitored (e.g. the price of gas), meaning that trends within different scenarios could be explicitly and realistically quantified. The response options evaluated were specific enough to be implemented on the ground, for example the development of emergency action plans with community involvement.

**Table Box 2.3:** The top five performing response options for the four scenarios. The response options were evaluated by participants using a numerical scale that yielded a combined score for total likelihood of adoption and total performance. (Modified from Aldrich et al., 2009).

| Response option | Procrastination blues | Stagflation rules | Nature be dammed! | Give rivers room! | Total |
|---|---|---|---|---|---|
| Hold regular, neighborhood meetings to "listen" to local adaptation needs, and mobilize local resources in response | 1 | 4 | 1 | 4 | 10 |
| Develop and update emergency actions plans with community involvement. Coordinate with State Emergency Management Office | 1 | 3 | 3 | 3 | 10 |
| Require local community governments to work with the NYS Emergency Management Office (NYSEMO) to complete and update regional hazard and pre-disaster mitigation plans | 1 | 3 | 2 | 3 | 9 |
| Require all state agencies to conduct flood audits of critical infrastructure | 0 | 3 | 3 | 3 | 9 |
| Change requirements for all new storm water permits | -1 | 4 | 2 | 3 | 8 |

Both technical and deliberative approaches to dealing with uncertainty in decision making often draw on the concept of ***adaptive management*** (Walters and Holling, 1990). Adaptive management is a formal procedure for learning by doing that is particularly amenable to sequential decision problems (Holling, 1978).

The sequential nature of the decision making is what provides the possibility for learning (from previous experience) and continuous improvement of future decisions resulting from a better understanding of (reduced uncertainty about) the system being managed. Adaptive management has seen strong application in fisheries (Hilborn, 1992; Walters, 2007), providing theory underpinning management strategy evaluation approaches (Smith, 1994). Terrestrial wildlife management and conservation have also seen the successful application of adaptive management (Johnson et al., 1997; McDonald-Madden et al., 2010), and its potential role in invasive species management is also recognised (Shea et al., 2002). Decision makers and policy analysts commonly invoke adaptive management as a valuable heuristic supporting continuous improvement, although many applications explicitly include a formal plan for learning (e.g. via model refinement); a fundamental aspect of adaptive management (Holling, 1978).

Despite the appeal of the adaptive management concept, documented examples of it working in practice are surprisingly few (Westgate et al., 2013, but see Box 2.4 for an exception). Successful working examples appear to be characterised by decision contexts involving a single jurisdiction, relatively few objectives to balance, the continuous involvement of strong technical expertise, relatively low social and cultural complexity and conflict, and a strong institutional commitment to ongoing management and funding to support it. Numerous reasons for the failure of adaptive management strategies have been proposed, including the failure to support ongoing monitoring and management costs.

**Box 2.4:** Dealing with uncertainty - adaptive management of North American Mallard ducks

Nichols and Williams, (2006) summarises an adaptive management programme that has been working since 1995 to support the management (hunting regulations) of mid-continent Mallard ducks (*Anas platyrhynchos*) in North America. The management objectives are to maximise the cumulative harvest over a long time period (including harvest devaluation when the predicted population size falls below the North American Waterfowl Management Plan goal threshold of 8.8 million breeding mallards). Management actions include four regulatory packages (intervention scenarios) that specify daily bag limits and season lengths for each of the four major North American flyways (Nichols et al., 2007). Four models of system response to harvest management are included in the model set. These models reflect two different hypotheses about the effect of hunting mortality on annual duck survival (compensatory mortality reflecting minimal effects of hunting and additive mortality reflecting maximal effects of hunting mortality), and two hypotheses about the strength of density-dependent relationships defining reproductive rates (weakly and strongly density-dependent).

At the initiation of this management process in 1995, all four models (representing all possible combinations of these four hypotheses) were given equal credibility weights of 0.25, indicating no greater faith in the predictions of one model than in those of any other (Figure Box 2.4). A complex monitoring programme is in place to estimate breeding population size and number of wetlands in Prairie Canada (an important environmental covariate), rates of survival and harvest, and pre-season age ratio. Each spring, the new estimate of population size is compared against predictions made the previous spring corresponding to each of the four models. These comparisons are combined with the model weights from the previous year to update the weights. Learning therefore occurs when weights become large for some models, giving them more credibility and thus more influence in the decision process, and small for others. The decision about which set of harvest regulations to implement depends on the system state, as defined by the estimated numbers of ducks and ponds.

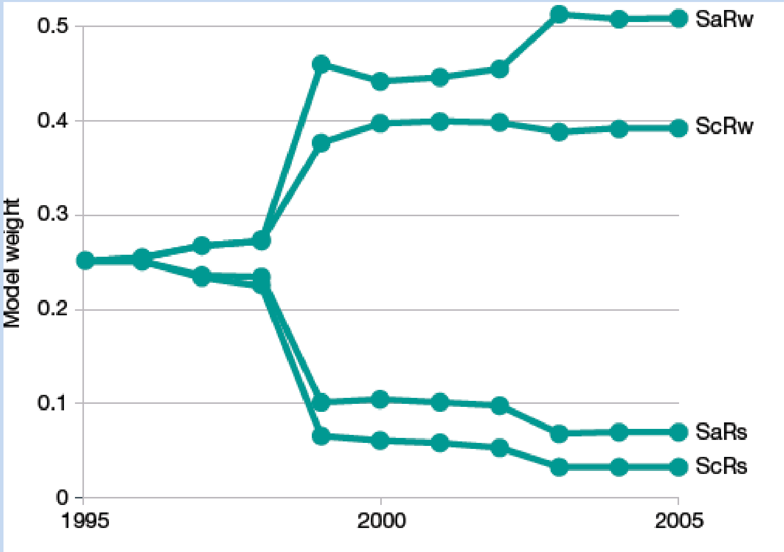


**Figure Box 2.4:** The evolution of belief for four models of Mallard duck responses to management (Modified by U. S. Fish & Wildlife Service, (2007)).

***Structured decision making*** (Gregory et al., 2012) is derived from MAUT (Raiffa, 1968; Section 2.3.2.1). However, structured decision making also draws heavily on more recent developments in decision analysis (Keeney, 1982; Hammond et al., 1998) and psychology (Kahneman and Tversky, 2000). It is an organised approach to identifying and evaluating creative options and making choices in complex decision situations.

Gregory et al., (2012) defines structured decision making as 'the collaborative and facilitated application of multiple-objective decision-making and group-deliberation methods'. Structured decision making is designed to deliver insight to decision makers about how well their objectives may be satisfied by potential alternative courses of action. It helps find acceptable solutions across groups, and clarifies divergent values that may underpin irreducible trade-offs. It is a very general approach to decision support (Figure 2.4), which can conceivably be applied to any environmental decision problem at any scale and any level of social and institutional complexity. It has the capacity and flexibility to utilise scenarios and models of almost any form to inform judgments about the implications for biodiversity and ecosystem services of any intervention or future. However, it is the value of structured decision making in situations in which there are conflicting values and conflicting views about the consequences of various courses of action due to uncertainty that differentiate it from the simpler analytical (or 'normative') approaches (Gregory et al., 2012). The attributes of structured decision making that distinguish it from MCDA are: the emphasis placed on understanding and dealing with difficult group dynamics through a collaborative, participatory approach to clarifying objectives; exploring cause and effect relationships; and dealing with contentious trade-offs. To some extent, the application of structured decision making formalises or prescribes an approach to dealing with the 'human' elements of decision making, including judgment bias, group dynamics and risk preferences. Tools such as MCDA may be used in a structured decision-making process where they add value or clarity to the process, but the process itself is not centred on the use of any such tool (e.g. Box 2.5).

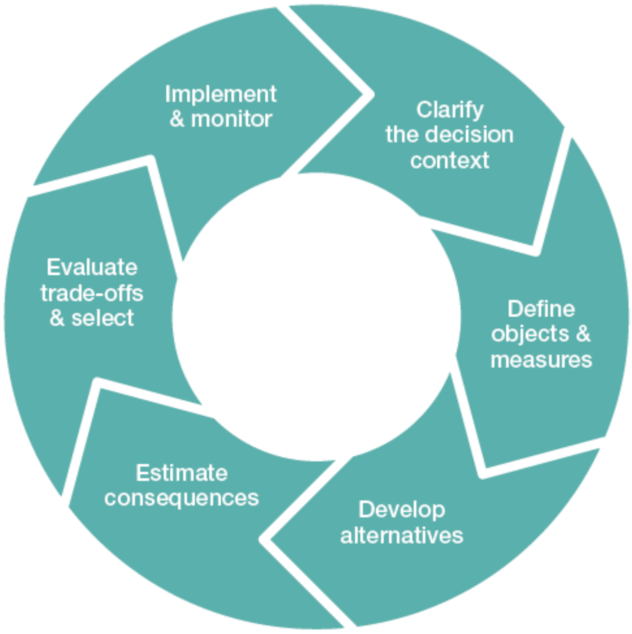


**Figure 2.4:** Six basic steps in structured decision making (Modified from Ecological Economics, 64/1, Failing et al., 2007, Integrating science and local knowledge in environmental risk management: a decision-focused approach, 47-60, copyright 2007, with permission from Elsevier). Note similarities with the policy cycle (Figure 2.1), adaptive management (Walters, 1986) and management strategy evaluation frameworks (Figure 2.5).

There are six basic steps identified in structured decision making (Figure 2.4; Gregory et al., 2012). Clarifying or scoping the decision context involves identifying what the decision is about, which decision or decisions will be made, by whom, and when. The spatial and temporal scale over which the decision applies is a key component of clarifying the decision context. Defining objectives and performance measures is a big focus of the structured decision-making approach, which defines what matters in the decision context and how these things will be measured. Objectives and performance measures drive the search for management and policy options and provide the basis on which they will be compared. The use of objective hierarchies is characteristic of most applications, possibly due to the strong focus on collaboration and encouraging participants to explore, and hopefully better understand, each other's values. Developing decision alternatives is a creative, deliberative process that aims to tailor candidate actions (or action sets) in a way that serves the defined objectives. Action sets can be thought of as intervention scenarios that can be played out in combination with exploratory scenarios about the future outside the control of decision makers. It is quite common that certain actions most suit the objectives of a particular stakeholder. Evaluating the performance of a particular stakeholder's preferred actions against the criteria of other stakeholders is a key part of understanding the consequences of each alternative. A basic tool used widely in structured decision making is the consequence table (Section 2.3.2.1), which sets out the expected outcome of each action for each performance measure relating to an objective. The process of estimating consequences of actions for objectives is a key place in which biodiversity and ecosystem service models can play a role in the approach. Models and scenarios can help in the exploration of expected outcomes arising from courses of action and the uncertainty about those expected outcomes. Evaluating trade-offs and selecting favoured options then proceeds by considering which options provide reasonable outcomes across all of the objectives considered. Proponents of structured decision making are generally eager to point out that the evaluation of trade-offs involves 'value-based judgments about which reasonable people may disagree' (Gregory et al., 2012). Finally, implementation and monitoring of the outcomes enables the post hoc evaluation of outcomes for the purposes of reporting and learning (McDonald-Madden et al., 2010), providing an opportunity for the structured decision-making process to be adaptive (Walters, 1986).

Two key strengths of structured decision making emerge from many of the reported applications. These include the clear separation of facts from values that is at the heart of the approach (Maguire, 2004) and the way in which the approach helps to partition and therefore simplify the technical and social complexity that commonly hinders most real-world decision problems.

One of the developers of structured decision-making theory and practice describes it as '... the formal use of common sense for decision problems that are too complex for informal use of common sense' (Keeney, 1982). This quote highlights the point that there is nothing mysterious or even particularly new about any aspect of structured decision making, other than the way in which it brings together many key concepts from decision theory to produce a workable protocol for deliberations.

A weakness of the approach is that guidance on how to undertake any given step within the 'cycle' tends to be minimal and vague. The key text on structured decision making for environmental applications (Gregory et al., 2012) emphasises that the use of the approach is something of an art. Knowing which specific tools to employ in any given decision context at each stage of the process requires significant experience, which means that the approach cannot simply be used 'off-the-shelf' by inexperienced analysts.

**Box 2.5:** Structured decision making for non-native fish management in the Glen Canyon Dam

Runge et al., (2011) describes a structured decision-making project run by the U.S. Geological Survey concerning the control of non-native fish below Glen Canyon Dam in the states of Utah and Arizona in the USA. They created a forum to allow agencies and tribes to articulate their values, develop and evaluate a broad set of potential non-native fish control alternatives, and define individual preferences on how to manage the trade-offs inherent in managing the problem. Two face-to-face workshops were held to discuss objectives and represent the range of concerns of the relevant agencies and tribes, and a set of non-native fish control alternatives was developed. Between the two workshops, four assessment teams worked to evaluate the control alternatives against an array of objectives (e.g. Figure Box 2.5). At the second workshop, the results of the assessment teams were presented. MCDA was used to examine the trade-offs inherent in the problem, and allowed the participating agencies and tribes to express their individual judgments about how those trade-offs should best be managed in selecting a preferred alternative. An effort was made to understand the consequences of the control options for each group's objectives. In general, the objectives reflected desired future conditions over 30 years. MCDA methods allowed the evaluation of alternatives against objectives, with the values of individual agencies and tribes deliberately preserved.

Trout removal strategies in particular parts of the catchment, with a variety of permutations in deference to cultural values, were identified as top-ranking portfolios for all agencies and tribes, based on cultural measures and the probability of keeping the endangered humpback chub (*Gila cypha* - www.iucnredlist.org/details/full/9184/0) population above a desired threshold. Sport fishery and wilderness recreation objectives were better supported by the top-ranking portfolio. The preference for the removal portfolios was robust to variation in the objective weights and to uncertainty about the population underlying dynamics over the ranges of uncertainty examined. A 'value of information' analysis (Runge et al., 2011a) led to an adaptive strategy that includes three possible long-term management actions. It also seeks to reduce uncertainty about the degree to which trout limit chub populations and explores the effectiveness of particular removal strategies in reducing trout emigration to where the largest population of humpback chub exist. In the face of uncertainty about the effectiveness of the preferred removal strategy, a case might be made for including flow manipulations in an adaptive strategy.

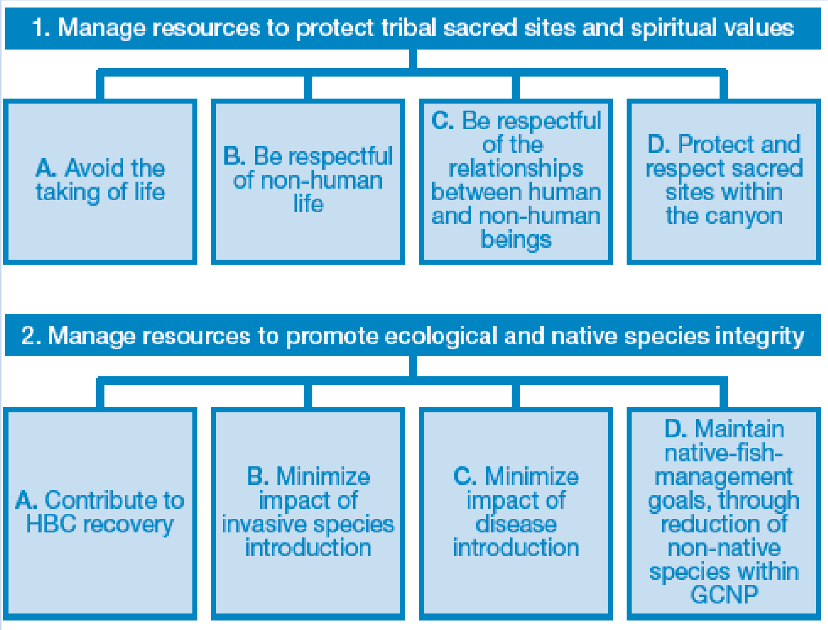


**Figure Box 2.5:** Example of hierarchies of two of the five fundamental objectives for non-native fish control below Glen Canyon Dam (Modified from Runge et al., 2011b, courtesy of the U.S. Geological Survey).

***Management strategy evaluation*** (sometimes termed *management procedure approach*, *harvest strategy evaluation* or *operating management procedures*) uses simulation models within an adaptive framework (Walters, 1986) to evaluate management options. The objective of the approach is to assess the consequences of alternative management strategies in a virtual world, taking multiple and often competing objectives into account (Butterworth, 2007; Bunnefeld et al., 2011; Smith, 1994).

Thus, management strategy evaluation can be used to reveal the trade-offs in performance across a range of management objectives (Holland, 2010; Smith, 1994). Management strategy evaluation does not prescribe an optimal strategy; instead, it provides the decision maker with information about the implications of different options (intervention scenarios) on which a rational decision can be based (Smith, 1994).

The conceptual framework and the subsystems modelled by management strategy evaluation are shown in Figure 2.5; the modelling steps are discussed based on Rademeyer et al., (2007). An 'operating model' (or, preferably, a set of candidate models) is created to address all of the key biological processes, trade-offs and uncertainties to which an ideal management procedure would be robust (usually one model is chosen as a reference model). These operating models (most typically population dynamics models) are used to compute how the resource responds to alternative scenarios (different future levels of catch or effort). The performance of each model is then integrated over all the considered scenarios. The likelihood of the occurrence of each scenario is regarded as a relative weight given to the output statistics. The final management strategy (procedure) is ideally chosen based on clear, a priori objectives.

Management strategy evaluation is typically used in the marine context to identify fishery rebuilding strategies and ongoing harvest strategies for setting and adjusting the total allowable catch, but terrestrial conservation applications are also likely (Winship et al., 2013; Edwards et al., 2014; Bunnefeld et al., 2011).

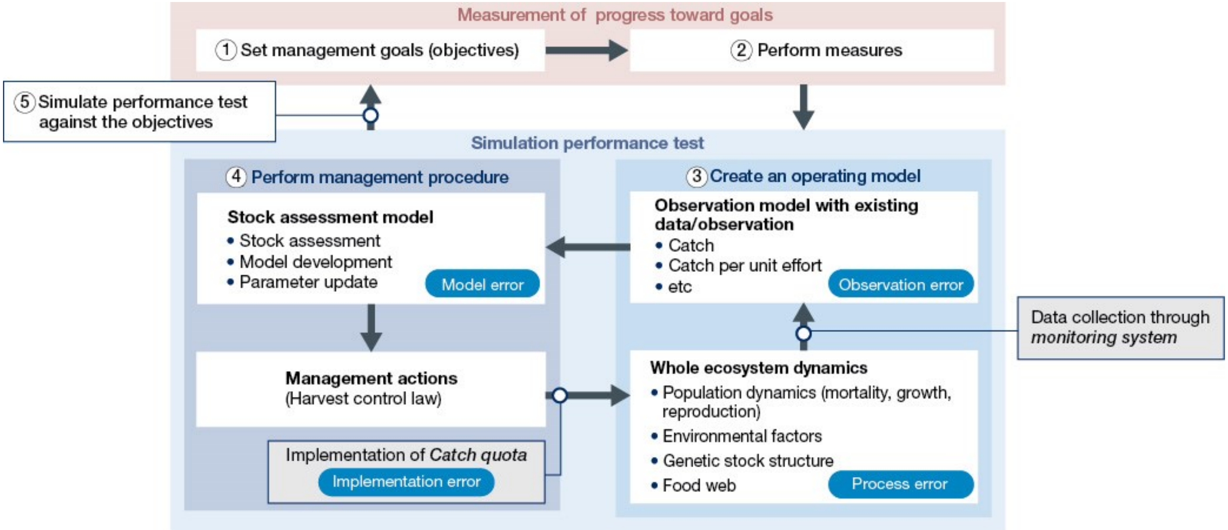


**Figure 2.5:** The management strategy evaluation framework (Modified from Adam et al., 2013, p.5). The top two boxes represent the management goals and performance measures used to measure progress toward those goals. An Operational model is created, which includes all the complexity of the ecosystem. Simulations of samples of that model is then performed which then feeds into stock assessment models. This procedure is performed multiple times, performing simulation tests which are used to evaluate how different management options 'perform', as measured by simulated outcomes for performance measures. The simulation performance test utilizes the models to simulate how management options play out under assumed ecosystem dynamics, how the outcomes of those options are measured and how those measurements are processed and interpreted through stock assessments to influence future harvest control settings. The MSE process effectively captures the process error (in the operating model), the observation error (in the observation model), the model error (in the stock assessment model) and the implementation error in the application of harvest controls.

A core strength of management strategy evaluation is its transparency and explicit consideration of natural variation and uncertainty in stock assessments and the implementation of management controls (Punt and Donovan, 2007; Holland, 2010). Multiple candidate models are generally considered within simulations to evaluate and test sensitivity to competing hypotheses (Rademeyer et al., 2007). Management strategy evaluation promotes consultation (Bunnefeld et al., 2011) whereby managers and other stakeholders can provide input into the candidate models and scenarios (Nuno et al., 2014), although participation is not a defining feature of management strategy evaluation. Recent applications have included indigenous interests in the management of socio-economic systems (Plaganyi et al., 2013), although technical demands due to complexity and reliance on computer simulation present challenges to its wider adoption in fisheries management (de Moor et al., 2011).

**Box 2.6:** Management strategy evaluation case study – joint management of fisheries in South Africa

Plagányi et al., (2007) reports on the management of South African sardine and anchovy fisheries. The two species have to be managed jointly as the anchovy harvest is necessarily accompanied by the bycatch of juvenile sardine; however, the latter is more valuable when adult, resulting in a trade-off. In the first joint management plan in 1994, total allowable catches were calculated based on abundance estimates from recruitment hydroacoustic surveys and spawning biomass. The total allowable bycatch of sardine was based on the anchovy total allowable catches, but the latter was not affected by the total allowable catches or the total allowable bycatch of sardine. However, the constraint posed by the sardine total allowable bycatch proved to be too strict, so that the management plan was updated in 1999 to allow a more flexible sardine total allowable bycatch to be set, depending on the relative recruitment estimations of the two species at any point in time. A trade-off curve was used in the selection of management goals to show explicitly the inverse relationship between the projected anchovy catch, with its associated juvenile sardine bycatch, and the directed (adult) sardine catch (Figure Box 2.6). Individual rights-holders in the fishery sector selected their own anchovy-sardine trade-off, rather than adopting a universal optimum. Recent recruitment estimates are based on an age-structured population model (de Moor, 2014). Early season catch quotas are tested by simulation to ensure robustness in terms of expected catches and uncertainties about the resource dynamics and harvest limits are adaptively adjusted during the year, as catch data are processed (De Oliveira and Butterworth, 2004).

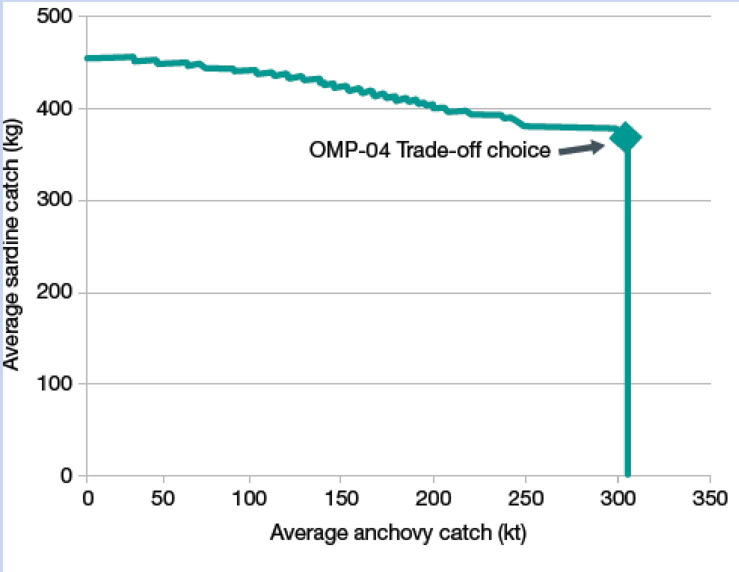


**Figure Box 2.6:** Trade-off curve between the average annual sardine and average annual anchovy catches, with the point selected for the 2004 operational management procedure (OMP-04) indicated. (Modified from Plaganyi et al. 2007, Making management procedures operational—innovations implemented in South Africa. ICES Journal of Marine Science (2007) 64 (4): 626-632, adapted and reused by permission of Oxford University Press on behalf of the International Council for the Exploration of the Sea. This image/content is not covered by the terms of the open access license of this publication. For permission to reuse, please contact the rights holder).

***Integrated territorial planning*** is a general and flexible approach to facilitate cooperative planning between neighbouring and sometimes overlapping jurisdictions and vertically from the individual land-use plot to the national and supranational levels.

The aim of territorial planning is to promote common interests or to reconcile objectives. Integrated territorial planning seeks to respond to jurisdictions that are recognised by specific national legislation and that are hierarchically organised, such as national, subnational, protected area, private and collective communal land (Amler et al., 1999). Applications of integrated territorial planning often include the establishment of multi-stakeholder platforms to facilitate spatial planning across areas that do not respond specifically to jurisdictions, such as watersheds, individual ecosystems or areas of influence of development projects. In this context, the strong links across the scales need to be considered in the analysis of land or marine area management (Ballinger et al., 2010). Integrated coastal zone management and integrated watershed management are examples of territorial planning in specific contexts that are implemented through cross-jurisdictional agreements between representative state, grass roots or private stakeholders (Alves et al., 2011; Ballinger et al., 2010). As integrated territorial planning tends to be GIS-based, its key strength lies in its visual products, including thematic maps that can be used across cultures, and its technical capacities to bridge knowledge systems by presenting both technical information and local knowledge and values.

The ***Delphi technique*** was developed by the RAND Corporation as a forecasting methodology (Gordon and Helmer, 1964; Linstone et al., 2002). Soon after, it was adapted as a decision tool (Rauch, 1979). Rauch, (1979) defines three relevant types of Delphi: classical Delphi, policy Delphi and ***decision Delphi***. The focus of classical Delphi is on forecasting and elicitation, or describing the future, while the latter two focus on mediating outcomes that influence the future. Classical Delphi may play a role in agenda setting, while policy Delphi and decision Delphi are particularly appropriate when decision making is required in a political or emotional environment, or when decisions affect strong factions with opposing preferences. Decision Delphi can be used formally or informally to exploit the benefits of group decision making while attempting to insulate against its limitations (e.g. deference to authority and groupthink). Example applications of Delphi as a decision tool include the allocation of national-level health funding in the USA (Hall et al., 1992) and setting priorities for the IT industry in Taiwan (Madu et al., 1991). Delphi can work as an informal, subjective decision-support model when the decisions are based on opinion, and can be converted to a formal model when quantitative data are available.

***Strategic Environmental Assessment*** (SEA) is the systematic environmental assessment of policies, plans and programmes (Therivel and Paridario, 2013). SEA can be viewed as a special case of environmental policy evaluation (Crabbé and Leroy, 2008) that falls within the broader field of policy evaluation, but that presents some very specific challenges due to the multiplicity of stakeholders' expectations concerning policies, and the political and thus debatable ground on which evaluations rest (Mermet et al., 2010).

SEA can be considered an evidence-based instrument that adds scientific rigour to policy development and implementation via suitable assessment methods and techniques (Fischer, 2007). SEA is not a decision-making tool, heuristic or framework in the sense of many approaches reviewed in this section that seek to identify best or robust decision or trade-offs (e.g. management strategy evaluation). It is an assessment process that provides information for planning, policy or programme development. The primary objectives of SEA (UNEP, 2002) include: i) supporting informed and integrated decision making by identifying the environmental effects of proposed actions, alternatives and mitigation measures; and ii) contributing to environmentally-sustainable development by providing early warnings of cumulative effects and risks that may not be apparent or may require assessment in individual environmental impact assessments (EIA: Du et al., 2012).

SEA is related to ***Environmental Impact Assessment*** (EIA), which is a widely used approach to evaluating the impact of projects (usually development proposals or other extractive or resource-use plans) on environments, including biodiversity and ecosystem services (Glasson et al., 2013). SEA and EIA are used at different levels of the decision-making hierarchy: while the former addresses policies, plans and programmes; the latter focuses on projects (Table 2.3). SEA tends to be more strategic and participatory, operating at higher levels in planning and programme development and being more forward-looking, potentially involving methods such as forecasting and visioning (Wang et al., 2006; Du et al., 2012; Liu et al., 2007).

**Table 2.3:** Summary of differences between SEA and EIA (Modified from sCBD and Netherlands Commission for Environmental Assessment, 2006).

| Strategic Environmental Assessment (SEA) | Environmental Impact Assessment (EIA) |
|---|---|
| Takes place at earlier stages of the decision making cycle | Takes place at the end of the decision making cycle |
| Pro-active approach to help development of proposals | Reactive approach to development of proposals |
| Considers broad range of potential alternatives | Considers limited number of feasible alternatives |
| Early warning of cumulative effects | Limited review of cumulative effects |
| Emphasis on meeting objectives and maintaining systems | Emphasis on mitigating and minimising impacts |
| Broader perspective and lower level of detail to provide a vision and overall framework | Narrower perspective and higher level of detail |
| Multiple processes, continuing and iterative, overlapping components | Well-defined process, clear beginning and end |
| Focuses on sustainability agenda and sources of environmental deterioration | Focuses on standard agenda and symptoms of environmental deterioration |

SEA is becoming more frequently and widely used (Fischer, 2007), with regulations and guidelines for SEA being proposed in many countries worldwide. For example, in the EU the SEA Directive (2001) requires an environmental assessment for plans and programmes at national, regional and local levels of jurisdiction. However, its role in assessing impacts of policies seems less well-developed. Increasingly, developing countries are introducing legislation or regulations to undertake SEA – sometimes via the modification of EIA legislation and policies (e.g. China, Belize, Ethiopia) and sometimes via natural resource or sectoral laws and regulations (e.g. South Africa, Dominican Republic). In Australia, 'strategic assessments' aim to analyse the cumulative impacts of multiple stressors on species listed as threatened under the Environment Protection and Biodiversity Conservation Act (EPBC, 1999). The CBD (Articles 6b and 14) (sCBD, 2005) encourages the use of SEA in its implementation (without making it a specific requirement). The Paris Declaration calls for the development of common approaches to environmental assessment generally, and to SEA specifically (www.oecd.org/dac). The CBD Conference of the Parties has endorsed guidelines for EIA and SEA (Decision VIII/28: www.cbd.int/decision/cop/?id=11042) and has also developed guidelines for their application in marine areas (Decision XI/18).

Primary strengths of SEA include the potential to integrate environment and development objectives, a reduction in the administrative burden of many small-scale impact assessments, and a reduction in the 'death-by-a-thousand-cuts' effect of many small impacts because of the explicit consideration of cumulative impacts at a regional scale (Hawke, 2009).

Other benefits include enhancing the role of science-based evidence in supporting decisions at higher strategic policy and planning levels than EIA, the capacity to identify and generate new options, the potential to build public engagement and improved transparency, an increased chance of early problem identification, the promise of transboundary cooperation, and clarity around institutional responsibilities.

However, SEA seems to lack an accepted underlying theory and the range of possible approaches that are described as strategic assessment appears almost infinite (Fischer and Seaton, 2002). It also lacks a standardised approach and therefore repeatability. While the intention is for SEA and EIA to work together in a hierarchy of tiered instruments (sCBD and Netherlands Commission for Environmental Assessment, 2006), with SEA taking place at a strategic level and EIA at a project level, the reality in some jurisdictions such as Australia is that large SEAs are replacing multiple, project-level EIAs (www.environment.gov.au/node/18607). This creates the real possibility, as well as the perception, that SEA provides an avenue for approval or endorsement of large impacts within a single assessment, which is viewed as negative by some stakeholders.

There are no hard rules about the nature of public consultation under SEA, which opens the method up to minimal or token consultation. Lack of expertise and specialist skills among the general public can lead to power differentials in the process where some stakeholders are well-resourced, informed and organised. In most administrations under severe human resource and financial constraints, SEA may be seen as a large administrative burden and impossible to properly manage, audit and enforce. The actual assessment of impacts or benefits on biodiversity and ecosystem services often defaults to the lowest common denominator; usually subjective risk matrix assessments and trend assessment. In 46 SEAs reviewed by the Organisation for Economic Co-operation and Development (OECD), the EU and the Japanese Ministry of the Environment (OECD, 2012; Sheate et al., 2001; Ministry of the Environment, 2003), all used subjective, largely data-free assessments of potential impacts on biodiversity and ecosystem services. This finding is supported by other comparative studies that find '… evolving SEA practice in Europe demonstrates the tendency to use the simplest available tools' (Dusik and Xie, 2009). Therivel and Walsh, (2006) observes that modelling has been little used among 200 United Kingdom authorities surveyed, and a survey of SEA practitioners in China found that 92% felt environmental mathematical modelling would be extremely useful, compared with around 60% who felt (risk) matrices, scenarios and expert judgment would be useful (YEPB and Ramboll Natura, 2009); yet not one mathematical model of biodiversity or ecosystem services is used in the sample of 15 case studies reviewed across Asia. It would appear therefore that the problem is the lack of available environmental models, not the lack of desire of practitioners to use them. The minimal role of modelling in current applications of SEA highlights an opportunity to expand its use and improve SEA.

**Box 2.7:** Strategic environmental assessment (SEA) of hydropower dams on the Mekong river

In 2009, the Mekong River Commission undertook an SEA of 12 proposed hydroelectric mainstream dams on the Mekong river to provide a broader understanding of the opportunities and risks of the development proposals. The Commission is in charge of implementing the 1995 Mekong Agreement for regional cooperation in the Mekong basin between the governments of Cambodia, Lao PDR, Thailand and Vietnam. These governments agreed on the joint management of their shared water resources to ensure sustainable development, utilisation, conservation and management of the Mekong river basin water and related resources. A number of independent environmental impact assessments (EIAs) that had been prepared in the lead up to the SEA were incorporated into the 'big-picture' framework of the SEA. The Commission was responsible for developing the strategic plan, alongside government agencies and experts, taking into consideration power security, economic development and poverty alleviation, ecosystems integrity, fisheries and food security, and social systems in the region.

After assessing the baseline status of the fisheries in the area, the potential impacts to both fisheries and the natural aquatic ecosystem functioning under different levels of damming were investigated. Five alternative development scenarios were developed by the Commission and compared to baseline statistics from 2000: i) a 'definite future' of already-approved dams; ii) no mainstream dams; iii) 6 (upstream) dams; iv) 9 (upstream and midstream) dams; and v) 11 dams. These scenarios represent clusters of projects with cumulative impacts on the Mekong river and surrounding areas. Employing

hydrological modelling forecasts, previous literature studies on the distribution and migratory patterns of Mekong river fish species and expert consultation and predictions of fishery yield impacts under the five damming scenarios were assessed (Figure 2.7). The analysis found that the mainstream projects would fundamentally undermine the abundance, productivity and diversity of the Mekong fish resources, as well as result in serious and irreversible environmental damage, losses in long-term health and productivity of natural systems, losses in biological diversity, and loss of ecological integrity. The SEA assessed four alternative courses of action for the immediate future and recommended that all further development of hydroelectric dams be deferred for a period of ten years. The strengths of this case study included the development of multiple, realistic, potential development scenarios; an extensive consultation process involving multiple governments, expert workshops and public involvement; and the evaluation of realistic, alternative courses of action. Extensive reporting is available at:
http://icem.com.au/portfolio-items/strategic-environmental-assessment-of-hydropower-on-the-mekong-mainstream/.

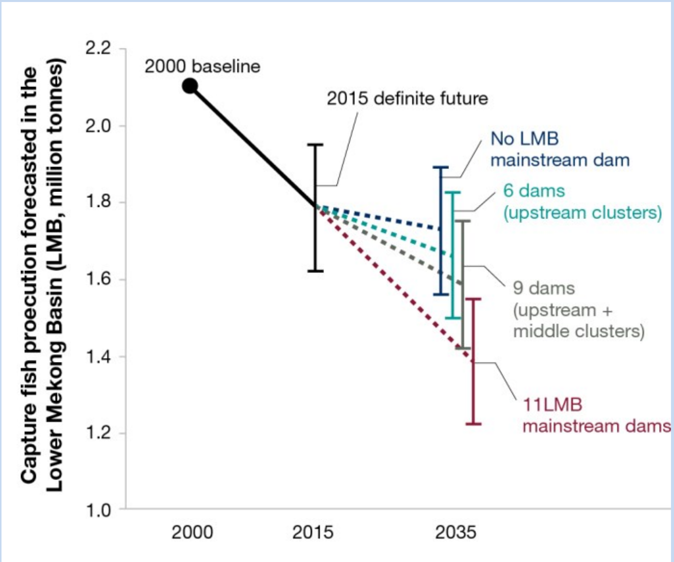


**Figure Box 2.7:** SEA of potential impact of mainstream dams on basin-wide fish production (Modified from ICEM, (2010)). Baseline fish production was anticipated to decline between 2000 and 2015 due to existing pressures on stocks. After 2015 a further decline was anticipated, but the magnitude of the decline depends on which dam building scenario is chosen, with the 11 mainstream dams clearly causing significantly greater reduction in fish stocks than the 'no further dams scenario', with other scenarios predicted to have intermediate impacts.

***The Open Standards for the Practice of Conservation*** were developed by Conservation Measures Partnership to provide a conceptual framework and specific tools for the successful implementation of conservation projects (CMP, 2013; Margoluis et al., 2013).

The 'open standards' refer to 'standards that are developed through public collaboration, freely available to anyone, and not the property of anyone or any organisation' (Dietz et al., 2010). The Conservation Measures Partnership, which was formed in 2002, is a consortium of non-governmental conservation and donor organisations. The Open Standards for the Practice of Conservation are a product of the Conservation Measures Partnership's mission to develop, test and promote conservation principles and tools that can credibly assess and improve the effectiveness of conservation actions (CMP, 2011). The Nature Conservancy Conservation Action Planning and the World Wildlife Fund Project and Programme Management Standards are similar endeavours to develop, adopt and implement standards for systematic project and programme management and monitoring (Moorcroft and Mangolomara, 2012; Margoluis et al., 2013). The Open Standards for the Practice of Conservation have been applied widely because of the large number of member organisations, and because training is provided by the Conservation Coaches Network (CCNet), whose regional franchises are increasingly serving as a mechanism to promote it globally (CMP, 2013).

Dietz et al. (2010, p.425) elaborated the five steps of the Open Standards for the Practice of Conservation, which are based on the project management cycle (Figure 2.6). The essential principles that apply to all of the steps include involving stakeholders, developing and cultivating partnerships, embracing learning, documenting decisions and adjusting as necessary (CMP, 2013). The standards are assumed to represent the 'ideal' conservation decision-making and learning process, but it is recognised that in reality standards can be implemented using a variety of tools and guidance, that few projects will start at the beginning of these standards, and that each project is different in potentially significant ways.

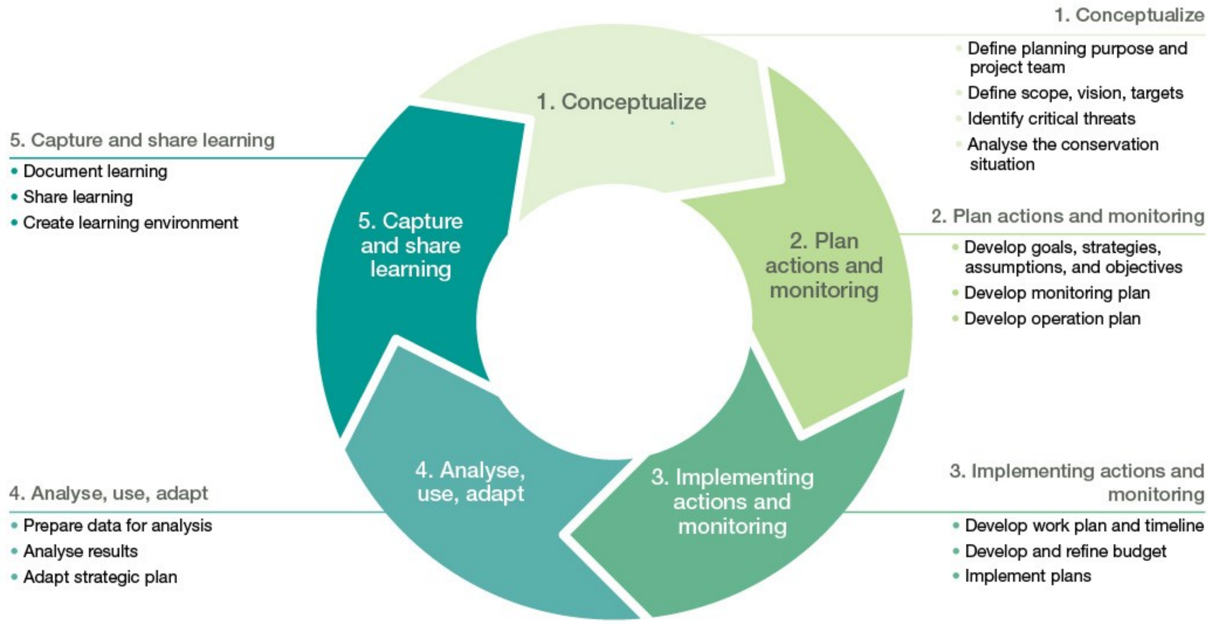


**Figure 2.6:** Five-step project management cycle of the Open Standards for the Practice of Conservation (Modified from CMP, 2013, https://creativecommons.org/licenses/by-sa/3.0/deed.en_US).

The Open Standards for the Practice of Conservation provide an overarching framework that can work with other conservation tools (e.g. Marxan, systematic conservation planning or structured decision making) (Schwartz et al., 2012). They are applicable at many scales, across different organisation types, and to different priorities within an organisation (Lamoreux et al., 2014). A key strength of the Open Standards for the Practice of Conservation is that they are supported by free software called Miradi Adaptive Management, which uses diagrams (e.g. results chains Salafsky, 2011), wizards, examples and multiple views. Miradi allows the practical and step-by-step application of the Open Standards for the Practice of Conservation framework (https://www.miradi.org/about-miradi/). As of 2012, Miradi had over 5,500 users in 167 different countries and had been used in over 115 projects of The Nature Conservancy (Schwartz et al., 2012). A criticism of the Open Standards for the Practice of Conservation is the lack of peer-reviewed publications that evaluate their effectiveness and place them in the spectrum of other decision-support and planning approaches (Schwartz et al., 2012). While the conceptual foundations of the Open Standards for the Practice of Conservation appear strongly connected to the policy cycle and adaptive management, it is hard to discern a particular theoretical foundation for the approach, for example compared with other approaches such as structured decision making in which each component appears to arise from sound decision theoretic foundations.

#### 2.3.2.4 Summary of strengths and weaknesses of decision-support protocols

The methods and approaches to decision support reviewed in the previous sections vary widely in their assumptions, strengths, weaknesses, complexity, sophistication and flexibility for dealing with the variability in decision contexts (Table 2.4). As with choosing between different types of scenarios and models, a key trade-off when choosing a decision-support approach is between simplicity (ease of use) and sophistication (the capacity to capture realism in terms of stakeholder perspectives, risk preferences, behaviour, and explicit, hidden and nascent objectives).

Clearly, using only consequence tables or risk matrices has simplicity on its side. Some expertise is required to step people through the process of using such tools properly, but the task is not overwhelming, which may explain why they are used so widely in SEA (Sheate et al., 2001; OECD, 2012). Many 'classical' decision-theory tools, such as the optimisation approaches, offer the allure of objective rationality. However, they do not perform well in isolation in many decision contexts because they fail to capture important aspects of human judgment and behaviour under risk and uncertainty (Kahneman and Tversky, 2000), and because socially important aspects of the decision problem commonly must be excluded due to technical constraints (the inability of optimisation software to cope with 'big' problems that involve many possible options, states of the world, and uncertainties). Nevertheless, when embedded as a component of a more holistic, deliberative decision process, classical tools such as optimisation may still make an important contribution to complex decision-making problems (Gregory et al., 2012). The key disadvantage of the more integrative approaches to decision support is the time and human resource overheads associated with running large, multidisciplinary, participatory approaches, and the capacity to alienate or generate cynicism if approaches are run poorly, or if there is a sense that stakeholder engagement is token. In short, big, complex decision problems with large consequences clearly demand sophisticated, integrated decision support, and shortcuts in either the technical or participatory aspects of these processes are taken at great peril.

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**Table 2.4:** Overview of assumptions, strengths and weaknesses of decision-support protocols described in this chapter, and extra case studies.

| Method family | Method | Assumptions | Strengths | Limitations | Case study | Case study and General reference |
|---|---|---|---|---|---|---|
| Methods tailored to multi-objective problems | *Consequence tables* | Implicitly assumes that users can coherently trade between consequences of actions across multiple objectives on a single arbitrary scale | Simplicity and usability, internal consistency, can utilize qualitative or quantitative assessment of consequences with respect to multiple objectives. Relatively easy identification of redundant and dominated options | Does not explicitly consider likelihood of outcomes. Does not provide very sophisticated approach to finding trade-offs. Time or risk preferences not explicitly incorporated | Water use planning in British Columbia, Canada. Consequence tables were used as part of a larger analysis into the allocation of water to hydroelectric dams, in the face of changing environmental and social values and knowledge of impacts. The consequence table was used to focus stakeholders on evaluating options and to make explicit the trade-offs between objectives | **Case study reference:** Failing et al., 2007 **General reference:** Gregory et al., 2012 |
| | *Cost-benefit analysis* | Benefits and costs of all values can be measured in a single currency (usually monetary). Preferences are revealed by observed actions or can be elicited | Relatively simple and transparent. Can use modelled or directly observed benefits and costs | Doesn't deal well with incommensurable values. Many stakeholders uncomfortable with monetarization | Planning options to mitigate losses due to coastal erosion in the coastal NSW, Australia were analysed using Cost-benefit analysis. The cost benefit analysis found that the most cost effective option was a 'Planned Retreat with Purchased Easements' option, which provides limited compensation for beachfront property owners in return for their agreement to vacate when trigger events occur | **Case study reference:** Balmoral Group Australia, 2014 **General reference:** Atkinson and Mourato, 2006 |
| | *Multi criteria decision analysis* | Explicit separation of cause-effect (likelihood and consequences) and value-based trade-offs. Estimates of likelihood and consequence can be qualitative or quantitative. Commonly uses multi-attribute value theory to describe preferences. Weights assigned using indifference methods | Weightings are a relatively simple way to express preferences for outcomes. Relatively sophisticated approaches exist for eliciting weights. Uncertainty about consequences can be incorporate. | Uncertainty about consequences difficult to incorporate. Most commonly uncertainty is characterised as probabilistic statements about likelihood (commonly under Multi-Attribute Value Theory approaches). | Using Multi Criteria Decision Analysis to reduce human-wildlife conflict in the UK. Two sets of stakeholders with opposing interests - raptor conservationists and grouse (a gamebird) managers - participated in a Multi Criteria Decision Analysis exercise to attempt to resolve their management conflicts. Setting explicit objectives and finding commonly favoured management strategies helped to build links between the two groups. | **Case study reference:** Redpath et al., 2004 **General reference:** Steele et al., 2009; Burgman, 2005; Diaz-Balteiro and Romero, 2011 |
| | *Analytic hierarchy process* | Variant of Multi-Attribute Value Theory designed to reduce elicitation burden. 9-point preference scales to translate ordinal judgements to cardinal judgements. Weights by pairwise comparisons | Relatively fast and simple to implement in a spreadsheet | Potentially susceptible to abuse and violations of basic decision theory axioms (but see mAHP approaches to addressing this problem) | Environmental Conflict Analysis in the Cape Region, Mexico. Model was utilized to determine suitability of land for different socio-economic activities and identify the land use pattern that maximized consensus among stakeholders from many different sectors | **Case study reference:** Malczewski et al., 1997 **General reference:** Mendoza and Martins, 2006 |
| | *Outranking* | Based on behavioural models of decision making. Sequential elimination of options. Weights assigned to objectives without considering range of consequences for each option | Easy to implement for decision problems with relatively many competing options. Existing user-friendly software. Can deal with qualitative or quantitative consequence estimates | Appears to fail on some basic axioms of rationality. Requires acceptances of behavioural models of decision making | Ranking the metapopulation extinction risk of a butterfly species in the Aland islands of Finland. Despite considerable occupancy data, there is high uncertainty in the metapopulation model outcomes (namely, extinction risk) for the Glanville fritillary. However, outranking provided a rank of management scenarios robust to the uncertainty in the model for this case study | **Case study reference:** Drechsler et al., 2003 **General reference:** Burgman, 2005 |

| Method family | Method | Assumptions | Strengths | Limitations | Case study | Case study and General reference |
|---|---|---|---|---|---|---|
| Optimization approaches | *Linear Programming* | Uses algorithms designed to optimise an objective function under specified constraints. Requires that a linear (or near-linear) relationship exists between actions and their expected consequences | Computationally quick. Linear relationships are easy to understand | Inappropriate in many ecosystems, where outcomes for objectives are non-linear in response to actions | Subsistence farming and the conservation of soil, to avoid erosion in the highlands of Ethiopia. Linear programming was utilised to determine the optimal production strategy to satisfy production goals while minimising soil erosion under a number of behavioural assumptions | **Case study reference:** Shiferaw et al., 1999 **General reference:** Chankong and Haimes, 2008 |
| | *Stochastic Dynamic Programming* | Able to incorporate complex, non-linear, dynamic relationships between management actions and outcomes. Accounts for stochastic events. Allows for optimal approach to differ depending on the state of the system | Allows complex, realistic relationships between actions and outcomes. Can incorporate effects of stochastic events | Requires highly detailed knowledge of cause-and-effect pathways in order to incorporate complexity | Optimal fire management to achieve ecosystem outcomes in South Australia. Stochastic dynamic programming methods were to provide state-dependent decision rules to managers about when to fight fires and when to let them run in order to achieve ecosystem composition objectives | **Case study reference:** Richards et al., 1999 **General reference:** Chankong and Haimes, 2008; Minas et al., 2012 |
| | *Goal programming* | Developed to handle multiple (usually conflicting) objectives, with each objective given a goal or target value to be achieved and deviations from the set of targets minimised | Avoids the naïve, binary-style step functions used by Linear Programming and Stochastic Dynamic Programming. Useful in single-stakeholder settings | Subject to misuse in multi-stakeholder problems through insincere goal-setting to manipulate outcomes | Optimising forest management for carbon sequestration in Spain. Goal programming was used to optimise the sometimes conflicting objectives of simultaneously maximising timber harvest and carbon sequestration through forest management practices. Though final solutions were robust to the weightings of the two objectives, they revealed the marked difficulty in obtaining both economic and forestry objectives | **Case study reference:** Diaz-Balteiro, 2003 **General reference:** Chankong and Haimes, 2008 |
| | *Structured decision making* | Provides insight to decision makers about how well their objectives may be satisfied by potential alternative courses of action. Assumes all relevant stakeholders can be identified and are willing and able to participate in process. Clarifies divergent values that may underpin irreducible trade-offs. Objectives and performance measures drive the search for management and policy options and provide the basis for comparison | Broad approach can be applied to a vast number of situations of different complexities and scales. Handles conflicting values and uncertainty well over more simple analytic tools. Partitions process into smaller steps, simplifying the social and technical components of the problem. Clearly separates facts from values | Large "human" element, so irresolvable conflicts between stakeholders can make progress impossible. Little guidance on how to undertake the steps within Structured decision making. Requires significant experience of Structured decision making to facilitate a fruitful outcome | Management of invasive willows in alpine Australia. Scientists worked with land managers to determine the best management strategy under a range of budgets to protect alpine bogs from willow invasion. The process involved developing a state-based dynamic model to describe the invasion of willows, role of wildfire in the ecosystem, and predict the effect of management interventions, as well as performing a value-of-information analysis | **Case study reference:** Moore and Runge, 2011 **General reference:** Gregory et al., 2012; Keeney, 1982 |
| | *Adaptive Management* | Sequential decision process, aiming to provide the opportunity to incorporate the role of learning into management programs. Operates on the idea of "continuous improvement" through flexibility in decision making, incorporating new information and re-assessing management actions | Enables the best decision to be made given the available information at a certain time, as well as improving on that decision (or strengthening certainty in the originial decision) over time as more information is gathered | Stakeholders often reluctant to commit to long-term monitoring of management outcomes (or unable to, due to unpredictable funding) | Adaptive management of sika deer populations in North Japan. Experts, hunters and government stakeholders identified four action plans for controlling over-abundant sika deer on Hokkaido. By varying harvest targets and improving estimation of population size, management was able to meet the objectives of maintaining the deer population at a level that posed little threat to crops and forests | **Case study reference:** Kaji et al., 2010 **General reference:** Oglethorpe, 2002; Salafsky et al., 2001 |

| | | | | | | |
|---|---|---|---|---|---|---|
| Integrative approaches (often subsume other approaches) | *Management strategy evaluation* | Simulation models within an adaptive framework for evaluating management options under conflicting objectives. Rather than prescribing an optimal strategy, Management strategy evaluation provides managers with the options and explicit trade-offs from which a rational decision can then be made. | Good at revealing trade-offs in performance across objectives. Explicitly and transparently considers uncertainty in both the assessment of the state of the environment and the implementation of management strategies. Encourages stakeholder input. | Technical demands in incorporating real-world complexity and reliance on computer simulations make Management strategy evaluation inaccessible. | Balancing fishery and conservation objectives in line fishing on the Great Barrier Reef, Australia. Stakeholders with competing objectives participated in a Management strategy evaluation to identify objectives and performance indicators of the system. Metapopulation and fishing simulation models were used to identify which often management scenarios maximised performance targets for the most stakeholders. | **Case study reference:** Mapstone et al., 2008 **General reference:** Bunnefeld et al., 2011; Holland, 2010; Rademeyer et al., 2007 |
| | *Scenario planning* | Multiple potential futures arise from a range of conditions and explicitly considering them may change current decisions. Development of futures can utilise qualitative and quantitative information. Does not asses probability of futures | Easily integrated into more complex decision support tools, such as Management strategy evaluation and Structured decision making. Promotes consideration of a wide range of possible futures, (rather than just likely ones); the assumptions behind them; and the uncertainties inherent in them | Doesn't account for, or enable assessment of, the probability of the futures. May therefore give too much consideration to highly unlikely events. Cannot accommodate/ predict futures that are not explicitly considered | Participatory scenario planning in the protected area of the Doñana social-ecological system in southwestern Spain. Stakeholders in conflict over the future of the region - for conservation versus development - participated in a scenario planning process to identify objectives for management of the protected area. Four futures were identified and backcasting was used to determine what actions and trade-offs stakeholders were willing to make in the present to secure the preferred future | **Case study reference:** Palomo et al., 2011 **General reference:** Börjeson et al., 2006; Chermack, 2011; Schoemaker, 1993; Schoemaker, 1995; Peterson et al., 2011 |
| | *Strategic Environmental Assessment* | Systematic and evidence-based assessment of environmental policies and programmes provides scientific rigour to the policy process and improves outcomes. Assumes assessment of projects at larger scales can provide better outcomes than multiple, smaller, individual actions | Facilitates integration of environmental and development objectives. Reduces both administration burden and cumulative impacts of multiple small projects by assessing policies at a regional scale. Encourages transparency and public engagement | Broad scales and numerous alternatives make for complex-to-analyse problems with high levels of uncertainty. Data collection difficult at such scales, often leading to further uncertainty. Public input may lead to reverse outcomes due to lack of expert knowledge and power differentials favouring well-resourced stakeholders. Often seen as infeasible under resource-stressed administrations | Assessment of cumulative impacts of housing, transport and mining infrastructure in the Perth and Peel regions of Western Australia, Australia. For a 8000km2 region of Western Australia, where urban expansion is planned to accommodation another 3M people, the biodiversity impacts of infrastructure were investigated as using distribution models of species' and communities' under 4 development scenario | **Case study reference:** Whitehead et al. 2015 **General reference:** Fischer and Onyango, 2012; Fischer, 2010 |
| | *Integrated Territorial Planning* | Requires cooperative planning between neighbouring/ overlapping jurisdictions and across multiple scales. Deals with jurisdictions that are recognized by specific national legislation and aims to promote common interests or reconcile conflicting objectives among these territories | Strong visual appeal as Integrated territorial planning often produces thematic maps that can be used to communicate across knowledge systems, bridging differences in culture and technical proficiencies. Incorporates both social and technical knowledge | Requires cooperation across administrative boundaries, which are often not designed to interact | Spain's Integrated Coastal Zone Management Strategy covers multiple territories across the country's coastline. Both sustainable development and integrated management objectives were developed to improve ecosystem health and socio-economic development goals. Coordination among the administrative levels is identified as integral to facilitating territorial coherence and achieving collective objective | "Report for Chapter VI of the Recommendation of the European Parliament and of the Council concerning implementation of Integrated Coastal Zone Management in Europe. http://ec.europa.eu/ourcoast/download.cfm?fileID=1323" |

### 2.3.3 Dealing with uncertainty in decisions

Uncertainty impacts on all phases of the policy cycle. In setting policy agendas, uncertainty may be invoked as a reason to pursue or avoid particular policies, or to motivate policy reform. For example, uncertainty about the magnitude of climate-change impacts on ecosystems and livelihoods may be used to invoke a precautionary approach to energy policy. Uncertainty impacts on policy design and implementation because, for example, there is often large epistemic uncertainty about benefits or impacts expected to arise from a particular policy or implementation strategy. Post hoc policy evaluation is often hampered by imperfect measurement of the outcomes of policy implementation that generates uncertainty around the evaluation of benefits and costs of policies and plans.

Uncertainties arise for a variety of reasons and take a variety of forms, some reducible and some irreducible (Wintle et al., 2010; Regan et al., 2002). Uncertainty can be addressed in a variety of ways during the agenda-setting and policy-design and implementation phases.

Exploratory scenarios provide an excellent means of characterising possible futures and exploring their implications (Chapters 1 and 3; Schwartz, 1995). A key step in scenario planning is the process of exploring the robustness of planning options to a broad range of possible futures (Schoemaker, 2012), which can be viewed as a heuristic for dealing with uncertainty.

Scenario planning can be viewed as arising from the discipline of future studies (Bell, 2003; Cook et al., 2014b), which subsumes a range of other agenda-setting and policy-support activities dealing with uncertainty about the future, such as horizon scanning (Sutherland and Woodroof, 2009), causal layered analysis (Inayatullah, 2004), visioning (Groves et al., 2002), emerging issues analysis (Molitor, 2003), backcasting (Robinson, 2003) and several others that cannot be described in detail here.

A multitude of mathematical methods exist for dealing with uncertainty in choice problems that can be useful in policy-design and implementation decisions. They include stochastic dynamic programming (Section 2.3.2.2), robust optimisation (Ben-Tal and Nemirovski, 2002), info-gap decision theory (Ben-Haim, 2006) and sensitivity analysis (Wallace, 2000).

An advantage of mathematical decision-support approaches for dealing with uncertainty is that the nature of the uncertainty being addressed is clear and precisely defined, as is the role of models. Models describe the system being managed and the nature and magnitude of the uncertainty. In addition to identifying robust options, the application of formal uncertainty analyses can highlight which uncertainties are most important to resolve and which are inconsequential (Moore and Runge, 2012). Such analyses provide a strong motivation and guidance for investing in the reduction of critical uncertainties. The primary impediment to the use of these approaches is the relatively high technical expertise needed and the limitations on the complexity and size of the decision problem that can be handled in practice.

Most of the commonly used mathematical approaches to characterising and dealing with uncertainty in decision making focus on epistemic and stochastic uncertainties, assuming rational, utility-maximising behaviour from decision makers. However, many environmental decision problems are characterised by high social complexity due to multiple stakeholders with diverse values operating in uncertain and shifting administrative, economic, political and legal environments (Balint et al., 2011). Such problems can seldom be fully characterised and analysed using mathematical approaches to uncertainty. A lack of specificity about objectives and a diversity in decision maker and stakeholder perceptions, knowledge, values and attitudes all introduce *decision uncertainty* (Chapter 1; Maier et al., 2008). Decision-support methods that address subjective and intangible uncertainties (e.g. Runge et al., 2011a; van der Sluijs et al., 2005) are therefore critical in supporting policy in most decision contexts. Such processes often require deliberation among decision makers and stakeholders to allow learning throughout the decision-making process. Participatory planning and adaptive management approaches exist that foster deliberation around epistemic, stochastic and decision uncertainty in ecosystem management (Susskind et al., 2012). A key challenge to dealing with uncertainty in participatory decision making is communicating to non-technical participants. A range of guiding documents exists to help provide a language for communicating uncertainty (Wardekker et al., 2008; Petersen et al., 2013) and to promote the use of graphical methods to help convey the nature and magnitude of uncertainty to a broader audience in a way that is more relevant to the decision at hand (McInerny et al., 2014).

## 2.4 Ingredients for success; matching approaches to decision needs

Reflecting on the sample of policy, planning and management support approaches and case studies reviewed here, a couple of observations about success emerge.

A defining feature of the documented successful applications of decision support appears to be the level of commitment and involvement of decision analysts or facilitators for the duration of the decision process.

Examples of decision processes were documented that ranged from highly participatory, deliberative, mostly non-technical exercises (Boxes 2.2 and 2.3), to more technical exercises (Box 2.4), and combinations of the two (Box 2.5). All had very strong commitment and support from decision analysts, modellers and/or facilitators. These people might be considered 'champions' of their given decision-support approach or method and, like champions of change, they are essential for the successful use of scenarios and models in formal decision processes (Guisan et al., 2013).

Section 2.3 described a sample of decision-support approaches and methods under broad families. Acknowledging that the approaches and case studies described in that section are based on a small sample of published applications in decision support (91 case studies in total), some generalisations are nonetheless supported regarding the sorts of decision approaches that lend themselves to application in particular decision-making contexts. While some aspects of this relationship between decision context and methods are self-evident – for example, the use of MCDA in decisions involving multiple stakeholders or decision makers – other patterns emerge which may be less obvious a priori. For example, it appears that – for the most part – sequential decision approaches tend primarily to address single-objective problems, while regional-scale, multi-objective problems tend not to be addressed using sequential, dynamic, adaptive management approaches (Westgate et al., 2013). This may be simply because regional-scale, multi-stakeholder decision problems tend to be one-off decisions with no plan or programme for future changes, or because the inherent complexity of such decisions precludes their analysis as sequential decision problems, even if they are so in reality.

Some lessons can be learned from the successful application of decision support, scenarios and models at multiple scales. At the global scale, Section 2.3.1 described an example in which scenarios of future land use (driven by consumption) and climate change, supported by a model that estimates the biodiversity outcomes of land-use change (GLOBIO), were used to motivate policy decisions, including the setting of Aichi biodiversity targets. This is not to claim that the targets themselves arose naturally from the prediction of a model, but simply that the analysis set the agenda by providing evidence of the scale of the problem and the consequences of not acting. Some attributes of this scenario and modelling work give some clues as to why this analysis had an impact on policy. Firstly, the scenario and modelling work was embedded in the institutional frameworks from which the relevant policies (e.g. Aichi biodiversity targets) arose. For that reason, the analysis had legitimacy and trust among many of the stakeholders and decision makers. Secondly, the work was timely and tailored to the policy problem. The evolution of the relevant policies has allowed sufficient time, and has been sufficiently transparent, that the analysis products could be well tailored to the policy needs. The analysis was at an appropriate scale, it analysed an appropriate range of scenarios, it provided outputs that were interpretable and motivating to policymakers, and it was credible – based on the best available science and modelling approaches at the time.

Reflecting on another example at a much finer scale, the Glen Canyon Dam non-native fish management problem (Box 2.5; Runge et al., 2011b) provides an excellent example of matching the decision framework, intervention scenarios and models of biodiversity impacts to a complex decision need. The problem involved multiple value and knowledge systems (Western and First Nations), multiple jurisdictions (USA governments and First Nations governments) and multiple sectors and stakeholders (military, wildlife management, water management, recreational fishers). This problem also involved high ecological complexity including introduced species, threatened species and a regulated river network. This decision context demanded a sophisticated decision-support framework, the dedication of decision-support facilitators, a tailored set of intervention scenarios, and a suite of models to describe the biodiversity and ecosystem service implications of different scenarios and decisions. Adopting a structured decision-making approach seems justified based on the decision context, and vindicated given the success of the biodiversity outcome and stakeholder acceptance.

Decision-support case studies utilising the methods and approaches described in Section 2.3 were categorised according to decision-context variables. An extract from that classification is provided in Table 2.5, with extra case study examples and guiding texts. While this table cannot be viewed as a comprehensive alignment of methods and decision contexts, it provides a framework in which to consider a choice of decision-support approaches.

**Table 2.5:** Decision-support approaches and methods assessed against the decision-context attributes defined in Section 2.2.1. Attributes are measured in the units described in Table 2.1. The extent to which a particular decision-support method or approach is relevant to a particular level of a context variable was subjectively assessed by the authorship group and invited experts. As such, these should be considered at best indicative of the context in which methods *have been* applied and not where they *could* be applied. The table presented here is a summary of the method-by-decision attribute spreadsheet developed by the authors for the IPBES scenarios and models expert group (Deliverable 3c) and the IPBES deliverable on policy and decision tools (Deliverable 4c).

| | | | Governance | | | | | Decision | | | Stakeholders | | Information | | | Ecology | | | | Scale | | | | | Case studies and references | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Method Family | Method | Policy cycle | Actors | History | Legitimacy | Sectors | Participation | Decision time horizon | Decision frequency | Objectives | Values | Knowledge system | Scientific knowledge | Data availability | Scientific capacity | Heterogeneity | Diversity | Flows across landscape | Stochasticity | Cross-scale dynamics | Temporal extent | Temporal grain | Spatial extent | Spatial grain | Case study | Case study reference | General reference |
| Methods tailored to multi-objective problems | Con-sequence tables | D,I | * | * | * | * | * | S | O | S | H | H | L | L | L | * | * | W | LP | W | * | * | * | * | Water use planning in British Columbia, Canada | Failing et al., 2007 | Gregory et al., 2012 |
| | Cost-benefit analysis | D,I | * | * | * | * | * | S | O | S | H | H | L | H | L | * | * | * | LP | W | * | * | * | * | Planning options to mitigate losses due to coastal erosion in the coastal NSW | Balmoral Group Australia, 2014 | Atkinson and Mourato, 2006 |
| | Multi criteria decision analysis | D,I | * | * | * | M | P | * | O | M | * | * | L | L | L | * | * | * | * | * | * | * | * | * | Using Multi Criteria Decision Analysis to reduce human-wildlife conflict in the UK | Redpath et al., 2004 | Steele et al., 2009; Burgman, 2005; Diaz-Balteiro and Romero, 2011 |
| | Analytic hierarchy process | D,I | * | * | * | M | P | S | O | M | * | * | L | H | H | * | * | W | LP | W | * | * | * | * | Environmental Conflict Analysis in the Cape Region, Mexico | Malczewski et al. 1997 | Mendoza & Martins 2006 |
| | Outranking | D,I | * | * | * | M | P | S | O | M | * | * | L | L | L | * | * | W | LP | W | * | * | * | * | Ranking the metapopulation extinction risk of a butterfly in the Aland Islands | Dreschler et al., 2003 | Burgman, 2005 |
| Optimization approaches | Linear programming | D, I | * | H | * | S | E | * | O | S | H | H | H | H | H | * | S | W | LP | W | * | * | * | * | Avoiding erosion in the highlands of Ethiopia | Shiferaw and Holden, 1999 | Chankong and Haimes, 2008 |
| | Stochastic dynamic programming | D, I | * | H | * | S | E | * | O | S | H | H | H | H | H | * | S | W | HU | W | * | * | * | * | Optimal landscape reconstruction for an endangered Australian bird | Westphal et al., 2003 | Chankong and Haimes, 2008; Minas et al., 2012 |
| | Goal programming | D, I | * | H | * | S | E | * | O | S | H | H | * | H | H | * | S | W | * | W | * | * | * | * | Optimising forest management for carbon sequestration in Spain | Diaz-Balteiro and Romero, 2003 | Chankong and Haimes, 2008 |
| aches) | Scenario planning | A,D,I | * | * | A | M | P | L | * | M | * | * | H | H | L | * | * | * | * | * | * | D | * | km | Participatory scenario planning in the protected area of the Doñana social-ecological system | Palomo et al., 2011 | Börjeson et al., 2006; Chermack, 2011; Schoemaker, 1993; Schoemaker, 1995; Peterson et al., 2011 |
| | Structured decision making | A,D,I | * | * | * | * | * | * | * | * | * | * | * | * | H | * | * | * | * | * | * | D | * | km | Management of invasive willows in alpine Australia | Moore and Runge, 2011 | Gregory et al., 2012; Keeney, 1982 |

| | | | | | | | | | | | | | | | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Integrative approaches (often subsume other appro | Adaptive management | D,I,R | * | * | * | * | * | L | R | * | * | * | H | H | H | * | * | * | * | * | * | D | L-R | km | Adaptive management of sika deer populations in North Japan | Kaji et al., 2010 | Oglethorpe, 2002; Salafsky et al., 2001 |
| | Management strategy evaluation | D,I,R | * | H | * | S | * | L | R | S | * | H | H | H | L | * | * | * | * | * | * | D | L-R | km | Balancing fishery and conservation objectives in line fishing on the Great Barrier Reef | Mapstone et al., 2008 | Bunnefeld et al., 2011; Holland, 2010; Rademeyer et al., 2007 |
| | Strategic environmental assessment | A,D,I | * | N | A | M | P | L | * | M | * | * | H | H | H | * | * | * | * | * | * | D | L-R | km | Assessing the impacts of future developments in the Perth-Peel region | Whitehead et al., 2015 | Fischer and Onyango, 2012; Fischer, 2010 |
| | Open standards for conservation | D,I,R | * | * | A | M | * | L | * | M | D | D | * | * | H | * | * | * | * | * | * | * | L-R | * | A plan to integrate landowner contact programs with landscape goals of blue oak woodland protection | Schwarz et al., 2012 | Schwarz et al., 2012 |
| | Integrated territorial planning | A,D,I | * | N | A | S | P | L | * | M | * | * | H | H | L | * | * | * | * | * | * | D | L-R | km | Spain's Integrated Coastal Zone Management Strategy | http://ec.europa.eu/ourcoast/download.cfm?fileID=1323 | |
| | | A=agenda setting, D=design, I=implementation, R=review | S=single, M=multiple | H=historical, N=novel | A=accepted, C=contested | S=single, M=multiple | P=participatory, E=executive | S=short-term, L=long-term | O=one-ff/once, R=repeated | S=single, M=multiple | H=homogeneous, D=diverse | H=homogeneous, D=diverse | H=High, L=low | H=High, L=low | H=High, L=low | S=single ecosystem, M=multiple ecosystems | S=single species, M=multi-species | W=weak, S=strong | LP=low/predictable, HU=high/unpredictable | W=weak external influence, S=strong | S=short-term, L=long-term | Seconds to millennia (Continuous scale) | L=local, R=regional, G=global | Meters (m) to kilometres (km) (Continuous scale) | | | |

An asterisk (*) indicates that the method is relevant at any level of the decision context attribute (e.g. can be applied at all spatial scales). An 'x' indicates that the method ignores the specific attribute (e.g. consequence tables do not consider uncertainty).

## 2.5 The role of scenarios and models in decision support

Decision-support needs of policymakers and managers are driven by the decision context (Section 2.4). The knowledge needs of a decision process are determined in the early phases of the process, including the decision-support framework and types of scenarios and models that will best satisfy those needs. The decision-support protocol chosen for a given decision problem determines whether, and which, scenarios and models can be used.

The capacity of different decision-support frameworks, protocols and approaches to utilise scenarios and models varies greatly. At one extreme, the simplest risk analysis approaches such as consequence tables (Section 2.3.2.1) can utilise model predictions of consequences for various objectives under candidate actions, but there is little scope within a consequence table to play out multiple scenarios about possible futures (although the candidate actions can themselves be viewed as simple intervention scenarios). In contrast, the more sophisticated, integrative approaches such as structured decision making, management strategy evaluation or SEA (Section 2.3.2.3) are amenable to utilising both intervention and exploratory scenarios and a great variety of models describing various aspects of biodiversity, ecosystem services and human behaviour.

Notwithstanding the role of scenarios and models generated outside decision processes for the purposes of improving knowledge and setting new policy agendas, the choice of scenarios and model outputs for a given decision or assessment should be determined by a clear articulation of the objectives of the decision or assessment. The importance of articulating clear and measurable objectives is emphasised in the decision science literature (Gregory et al., 2012), and a number of tools exist to help articulate objectives, such as objectives hierarchies. The most common problem that arises when choosing scenarios and aligning the outputs (response variables) of models with the fundamental objectives of an assessment or decision, is that objectives have not been clearly articulated, or they are embodied in vague statements such as 'ensuring a sustainable future for municipality *x*', for which there may be a huge set of relevant model outputs or indicators.

Another common impediment to choosing model outputs that are proximal to fundamental objectives is that the measures prescribed in the objective statement are impossible or highly impractical to model. For example, a regional level objective to secure all remaining mammals in the Amazon basin while increasing economic opportunities for local peoples could conceivably be supported by models of the population viability analyses of all mammals and socio-ecological models of local livelihoods under a range of future climate, land-use and intervention scenarios (e.g. Wintle et al., 2011). However, it is highly unlikely that population viability analyses for every Amazon basin mammal could be constructed in time to influence any decision process. For this reason, surrogate model outputs that can be developed within time, budgetary and expertise constraints are commonly used to approximate the ideal measure of the fundamental objective.

For terrestrial ecosystems at almost all spatial scales, one of the most commonly used surrogates for biodiversity are species distribution models and various aggregations of those models. Species distribution models are appealing for the reason that observation data and mapped environmental variables are readily and freely available, the technology to fit and evaluate species distribution models is readily available and easy to use, and large numbers of species can be processed rapidly. However, with the many benefits of distribution modelling come many limitations and drawbacks that are well documented (Zurell et al., 2009; Fordham et al., 2011; Guisan and Thuiller, 2005). For the purposes of characterising and predicting long-term biodiversity persistence they are a useful, if blunt instrument. They do not adequately characterise many of the spatial and temporal processes that mediate persistence or extinction in changing environments. For example, failure to explicitly deal with dispersal limitations, competition and predation, or the plasticity and evolution of thermal and other niches, means that they may be missing much that is important in the extinction process. However, by representing spatial and temporal variation in the availability of suitable habitat, they provide a distal surrogate for species persistence over medium to long time frames. Combined with other coarse analyses (e.g. Carroll et al., 2010) that consider spatial processes, they may provide useful information about the relative merits of alternative conservation options under environmental and land-use change. While it is desirable for model outputs to directly reflect the fundamental objectives of a given assessment of a decision problem, in many instances it will not be possible, and in such instances surrogate outputs (e.g. species distribution models) are often better than nothing. Careful consideration of exactly what value model outputs bring to an assessment or decision problem is therefore a necessary ingredient for successful integration at the decision/modelling interface (Addison et al., 2013).

The need for scenarios is generated at the 'assessment and decision-support interface' (see Figure 1.3 in Chapter 1), based on the assessment/policy/planning/management problem at hand. For example, if a coastal management body needs to make decisions about where to allow housing development, it may need carbon emissions scenarios to underpin modelling of sea-level rise in its region and to characterise uncertainty about future sea levels relating to emissions. A regional biodiversity assessment may require human population growth and land-use change scenarios to inform biodiversity models used to make projections of biodiversity change over several decades (Bomhard et al., 2005). Two broad classes of scenarios are described in Chapter 3: exploratory scenarios and intervention (target-seeking and policy-screening) scenarios. Exploratory scenarios are used to explore the sensitivity of response variables (e.g. species persistence or freshwater availability) to a range of possible futures. How exploratory scenarios are determined and how many can or should be considered is open to the interests, concerns and imagination of the participants in any given assessment or decision problem. Scenarios can be generated by asking 'What future contingencies are likely to impact on the environmental assets, goods and services from our region that we value?' or through more formal or structured means of scenario elicitation (Carpenter et al., 2006).

Exploratory scenarios have been applied in many types of assessments at all spatial scales, from local to global (Alkemade et al., 2009; MA, 2005). Intervention scenarios represent possible or anticipated futures arising under a set of specified interventions. Interventions can take the form of policy options, planning options or management actions and should be specified within realistic social and economic constraints so that they can be considered plausible futures given a certain policy pathway (e.g. Sandker et al., 2009). While intervention scenarios fit most naturally in the domain of decision analysis, they also play a role in policy design and implementation (see Chapter 1).

## 2.6 Barriers and knowledge and capacity-building needs

The ingredients for the successful use of decision-support frameworks, scenarios and models in decision making are often missing in big environmental decision problems, creating barriers to adoption. A key ingredient that can be hard to obtain in decision problems is the dedication and continuity of involvement of decision-support facilitators and modellers in close collaboration with decision makers throughout the decision-making processes.

There is a mismatch between the preponderance of academic and theoretical studies around scenario development, modelling and decision-support approaches, and the relatively small number of documented case studies that present the successful application of scenarios and models in decision making in the environmental sector. This is especially the case at the broader regional and global scales.

It is hard to imagine that the relatively small number of documented successful examples of modelling and scenario analysis in decision making is due solely to a lack of champions. Examples of the successful application of formal decision approaches such as MCDA and scenario planning (often using scenarios and models) abound in other sectors such as manufacturing, business and the military. However, there appears to be a particular impediment to the wider application of such approaches in biodiversity and ecosystem service policy design and implementation. This may relate to the complexity of socio-ecological systems, a general lack of trust in data and measurement methods, a lack of good quality data, a lack of willingness to invest in collecting good quality and relevant data, or a lack of willingness to invest the time and financial resources necessary to ensure the successful application of scenario analysis, modelling and decision support in environmental decision problems.

Access to relevant data and models is an issue recognised at all scales (Dusik and Xie, 2009). This issue can be partly addressed by data and interface development (addressed in Chapters 7 and 8), although the reality is that insufficient financial resources are allocated to collecting data and building models relevant to most decision problems, irrespective of the magnitude of potential impacts.

For example, an assessment of the potential impacts of 11 new dams on the Mekong river involved almost no new data collection, but instead relied on the synthesis of sparse existing data (Box 2.7). For this reason, model outputs often fail to meet decision-making needs. This can be partly addressed by improving communication and expectations about the capacity of models to deliver the information relevant to decisions, and by improving investment in data collection and the capacity of models to deliver what is required, through training, technical advances and the standardisation of best practices (Peer et al., 2013).

A lack of appreciation of the potential role of decision support, scenarios and models on behalf of decision makers is another impediment to uptake. This appears to be partly due to a lack of trust in modellers, models and scenarios, and partly due to a lack of education about the potential benefits of decision-support tools, which may be due to a lack of exposure to working examples that highlight the benefits to decision makers of engaging with decision-support tools and practitioners.

This problem can be exacerbated by a lack of data to underpin the models and scenarios of most interest to policymakers and managers, a lack of willingness on the part of modellers to engage fully in real-world decision problems and develop the most relevant scenarios and models for the problem at hand, a lack of transparency in approaches to modelling and scenario development, and complex political agendas that are not amenable to the transparency ideally associated with good modelling and scenario analysis.

A subset of these problems can be overcome through improved communication and better documentation of the successful application of scenarios, models and decision support (Gibbons et al., 2008). The exploration of methods to improve the credibility of model predictions through the collection of empirical evidence demands further attention and resources. The capacity of models to sensibly characterise uncertainty is a key component of their credibility, indicating an important area of research and development in modelling research. Increased collaboration between modellers and decision makers will lead to increased trust, better and more relevant scenarios and models, and a culture of decision support based on scenarios and models that is robust to complex political agendas.

Capacity in scenario analysis and modelling varies geographically. In relatively wealthy countries, scenario development and modelling skills among environmental professionals are low relative to the number of assessment and policy implementation processes that would benefit from the injection of such skills. This problem is magnified in developing countries where, arguably, there is weaker environmental governance, more pressing environmental impacts, and less resources available to address them, including resources invested in scenario analysis and modelling skills.

The challenge of increasing uptake of decision-support approaches is, in part, a cultural one. The capacity of modellers and decision analysts to influence decision processes in a positive way is impaired by communication challenges across disciplinary divides, and the fact that much of the skill base resides in academic institutions, for which there are few tangible rewards for being involved in real decision processes.

Making scenario development, modelling and decision processes genuinely participatory brings cultural challenges and benefits. A key benefit is that participatory decisions and plans are more likely to be accepted and adopted by those who feel empowered through participation in the decision process.

Cynicism about scenarios, modelling and decision support may partly exist due to a sense among the general public and stakeholders that these tools are used by authorities and experts to maintain, rather than share, power. This attitude may exist partly because of a lack of genuine participatory modelling, scenario and decision processes, and could be partly mitigated by increasing the prevalence of genuinely participatory scenarios, modelling and decision processes. There may be a role for IPBES in facilitating and promoting a network of participatory scenario, modelling and decision-support practitioners to build capacity globally.

Cultural challenges extend to negotiating political forces that may not be completely comfortable with 'handing over' complex and sensitive decisions to technocrats using systems that policymakers do not fully understand or trust, or that may be uncomfortable with the level of transparency about motivations, values and scientific facts that decision support brings to decision making. This implies several key challenges. There is the challenge of educating policymakers to understand that involvement in decision processes does not have to mean relinquishing power. Convincing policymakers to engage with decision support requires conveying the notion that decision support can judiciously utilise models and scenarios, can help reduce complexity, distil true differences of opinion and values from linguistic ambiguities or confusion, increase mutual understanding of each other's values, and reduce conflict.

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# 3 Building scenarios and models of drivers of biodiversity and ecosystem change

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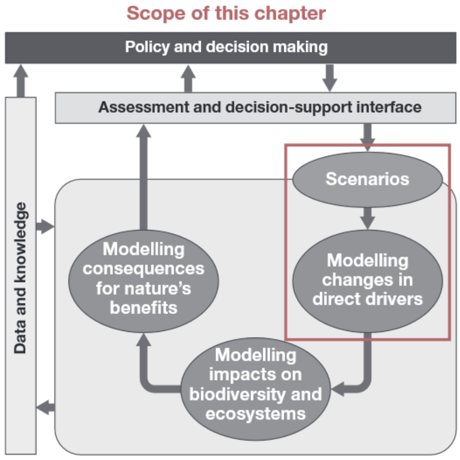
# Table of contents

**Purpose of this chapter:** Provides an overview of broad types of scenarios for addressing the various policy and decision-making contexts introduced in Chapter 2; and critically reviews major sources of scenarios of indirect drivers and approaches to modelling resulting changes in direct drivers that can, in turn, serve as inputs to modelling impacts on biodiversity and ecosystems (covered in Chapter 4).

**Target audience:** A broader, less technical audience for the overview of scenario types; but a more technical audience for the treatment of particular scenario and modelling approaches.




# Key findings

**Expert-based and participatory methodological approaches to scenario development represent different sets of tools with respective advantages and disadvantages (3.2.1).** Expert-based approaches are ideal during assessments in which empirical data can provide a solution and formal modelling is necessary. Expert-based methodologies are also appropriate for developing scenarios and models of indirect drivers, particularly as the temporal and spatial scales as well as uncertainties increase. Participatory approaches are ideal when dialogue among local stakeholders is key to successful assessment outcomes as well as when local and indigenous expertise can supplement scientific knowledge at the spatial scale under consideration. Local ecological knowledge is valuable when assessing drivers at local spatial scales as a complement to other expert-based methodologies, particularly within the context of assessment resource and time constraints.

**Choice of the type of scenario – exploratory or intervention – is highly contingent on the policy cycle decision-making context (3.2.2).** Exploratory scenarios are most often utilised during the initial problem identification stages to allow for the projection of multiple possible futures as well as the identification of relevant stakeholders and problem specificities. While also employed in direct driver scenarios (scenarios of drivers), exploratory scenarios are particularly pertinent to investigating scenarios of potential indirect drivers. Intervention scenarios and techniques such as backcasting for target-seeking scenarios are more useful in later stages of the policy cycle where there is a consensus on the desired goals and the focus is on potential pathways to such goals. Ex-ante (policy screening) and ex-post (retrospective policy evaluation) assessments are mutually reinforcing and complementary approaches in the policy cycle, and scenarios are very useful tools supporting these assessments.

**No single model of drivers of change in biodiversity and ecosystem services can capture all dynamics at a high level of detail (3.2.3).** The coupling or integration of models has become an important tool to integrate different scales and dimensions. Treatment of the spatial and temporal scales at which drivers operate as well as their interactions is crucial for the construction of consistent and comprehensive scenarios on biodiversity and ecosystem services. Complex models can coexist with and be complemented by more stylised and simplified models. Stylised models can be useful to identify simple tipping and reference points.

**Indirect and direct drivers interact on various spatial, temporal and sectoral scales, producing synergies and feedbacks that need to be taken into consideration. Failure to consider such dynamics**

**can potentially render scenario analysis incomplete, inconsistent or inaccurate (3.3, 3.4).** Prominent indirect drivers exhibit significant interlinkages among themselves as well as with direct drivers of biodiversity and ecosystem change. Due to the nature of sociocultural phenomena, certain indirect drivers and their interlinkages are particularly difficult to explicitly formally model, yet need to be represented in scenarios of indirect drivers (3.3). As with indirect drivers, direct drivers also display considerable interlinkages and feedbacks, with significant potential for cascading effects on biodiversity and ecosystems (3.4).

**Existing scenarios can serve as useful points of departure but are not likely to be appropriate in terms of temporal, spatial and sectoral scales and may not contain sufficiently detailed storylines to be useful for the construction of Intergovernmental Platform on Biodiversity and Ecosystem Services (IPBES) driver scenarios (3.5).** Scenarios need to be specifically tailored to the context of the biodiversity and ecosystem services studies. In many cases, the environmental scales (e.g. habitats, biomes) may be more relevant for IPBES driver scenarios than institutional scales (e.g. administrative, municipal, provincial, country). Existing scenarios can be useful for the information they contain, but typically provide limited insight if applied without proper adaptation to the decision context of a particular biodiversity and ecosystem study.

## Key recommendations

**IPBES is encouraged to adopt tailored driver scenario methodologies reflecting the requirements of a biodiversity and ecosystem services-specific decision-making context (3.2).** Participatory modelling approaches are ideal in situations where local stakeholder involvement and collective governance are key to developing planning pathways, while expert-based approaches are best utilised when formal modelling methods and more rigorous quantitative analyses are required. Exploratory scenarios are best utilised in the initial policy cycle phases to elucidate potential futures of indirect and direct drivers. Intervention scenarios, in particular target-seeking scenarios, are advantageous later in the policy cycle to formulate more concrete planning pathways for achieving goals associated with direct drivers. Indigenous and local knowledge is crucial for understanding the nature of the various drivers and the richness of their interactions in specific contexts.

**IPBES is encouraged to invest in the development of and capacity building for the modelling of drivers (3.3, 3.4).** The IPBES Task Force on Knowledge, Information and Data and the follow-up activities of the scenarios and modelling deliverable are encouraged to facilitate the improvement of tools to integrate across scales. In order to broaden the capacity to create and use these tools, the Task Force on Capacity Building would benefit from a specific focus on making these tools more freely available and on training programmes. Spatially nested modelling approaches of indirect and direct drivers would be ideally employed to construct globally-consistent national/local driver analysis. Driver scenarios need to address all relevant drivers of biodiversity and ecosystem services and connect short-term phenomena with long-term trends.

**IPBES deliverables dealing with scenarios and models, in particular author teams of the chapters on drivers of biodiversity and ecosystem change in IPBES regional assessments, are encouraged to carefully explore the interactions among indirect and direct drivers (3.3, 3.4).** An improved understanding of potential driver synergies and feedbacks on the various spatial, temporal and sectoral scales is essential to the construction of biodiversity and ecosystem services-specific scenarios and models. This analysis is particularly relevant for assessing the extent to which findings and conclusions on drivers at a specific scale may be relevant for extrapolation to other scales.

**IPBES is encouraged to develop new scenarios of indirect and direct drivers that provide added value compared to existing global environmental assessment scenarios such as the Intergovernmental Panel on Climate Change (IPCC) Shared Socio-economic Pathways (SSPs)/Representative Concentration Pathways (RCPs) and scenarios developed for the Millennium Ecosystem Assessment (MA) (3.5).** While existing global scenarios can serve as reference points against which to benchmark specific IPBES driver scenarios, collaboration with other scenario development activities outside of IPBES (e.g. under the IPCC) is seen as highly beneficial. However, IPBES requires novel scenarios that address those direct and indirect drivers relevant to biodiversity and ecosystem services at spatial and temporal scales relevant to the underlying processes involved. Scenario development would benefit from reducing inconsistencies and fostering greater creativity within scenario storylines to capture the possible development directions of the multiple drivers underlying biodiversity and ecosystem services.

## 3.1 Introduction

Ecosystems and biodiversity have been influenced by natural drivers of change ever since the beginning of life on Earth. Until human activities began exerting considerable ecological impacts, ecosystems and biodiversity evolved under the influence of natural drivers such as changing climatic and lithospheric conditions. Drivers associated with human activities (anthropogenic drivers) have accelerated the rate of species extinction and significantly altered ecosystem properties to the extent that less than 25% of the remaining land surface remains ‘natural’ (Ellis, 2011). Some scientists have proposed naming this new geological epoch the Anthropocene, in which human activities in recent centuries have become the dominant drivers of change in the Earth’s atmosphere, lithosphere and biosphere (Crutzen, 2006). There is now growing evidence that local-scale forcings (e.g. land-use change) may lead to a threshold-induced state shift with significant implications for the Earth’s biosphere (Barnosky et al., 2012).

Chapter 3 focuses on approaches to building scenarios and models of drivers, and therefore provides a link between the policy and decision-making context elaborated upon in Chapter 2 and the modelling of impacts of these drivers on biodiversity and ecosystems covered in Chapter 4 (see Figure 3.1) and, in turn, on nature’s benefits to people (including ecosystem services) and human well-being in Chapter 5. Chapter 3 builds on the discussion in Chapter 2 of policy and decision-making needs relating to different phases of the policy cycle, by providing an overview of methodologies for building scenarios and models of indirect and direct drivers to address these needs. The chapter begins with an examination of methodological approaches, including participatory and expert-based methods for developing scenarios, followed by a summary of scenario types employed in the field of environmental assessments and decision making. The uses and implications of several scenario approaches as well as ex-ante and ex-post assessments are explored (see Section 3.2.2.3). Modelling methods and the linkages between models are presented, followed by detailed overviews of prominent scenarios and models of indirect and direct anthropogenic drivers. The chapter concludes with an examination of the research needs and gaps that need to be addressed as biodiversity and ecosystem services assessments progress.

### 3.1.1 Definition and classification of direct and indirect drivers

Scenarios of change in drivers are a basic component of models projecting biodiversity and ecosystem change.

Indirect drivers are drivers that operate diffusely by altering and influencing direct drivers as well as other indirect drivers (also referred to as 'underlying causes') (MA, 2005b; sCBD, 2014).

Understanding the role of indirect drivers is vital to understanding biodiversity and ecosystem change at the direct driver level. Indeed, indirect drivers frequently have primacy within the causal framework linking drivers to biodiversity and ecosystem change. Indirect drivers considered in this assessment include economic, demographic, sociocultural, governance and institutional, and technological influences.

Direct drivers (natural and anthropogenic) are drivers that unequivocally influence biodiversity and ecosystem processes (also referred to as 'pressures') (MA, 2005b; sCBD, 2014).

Over a long enough time frame, the impacts of direct drivers of change in biodiversity and ecosystem services nearly always influence anthropogenic indirect drivers, thereby resulting in feedbacks between direct and indirect drivers (e.g. economic implications of climate change, overexploitation, and habitat modification on global fisheries (Sumaila et al., 2011). Furthermore, many direct drivers interact with other direct drivers, highlighting the complex interlinkages that need to be taken into consideration throughout assessment analyses. This chapter specifically examines the following direct drivers: land-use change, climate change and pollution, natural resource use and exploitation, and invasive species. Indirect drivers also contribute to anthropogenic assets in the form of infrastructure, knowledge, technology and financial assets. Anthropogenic assets result from the interaction between society and nature and contribute to human well-being, although their relative importance is context-specific.

Drivers are not to be viewed as separate, static influences, but rather considered as dynamic factors interacting with and within each other. Indirect drivers frequently strongly interact, giving rise to complex emerging properties on various spatial and temporal scales.

### 3.1.2 Chapter overview

As elaborated upon in Chapter 2, stages of the policy cycle range from agenda setting to policy implementation and eventual review. The policy cycle serves as a framework to facilitate effective decision making by taking into consideration a comprehensive analysis of the problem, followed by policy design, implementation, and finally evaluation of policy impacts. Accordingly, the specific policy and decision-making context of any given assessment of biodiversity and ecosystem services will to a large extent determine the point of departure for subsequent methodological approaches to building scenarios and models of drivers (see Figure 3.1). Participatory and expert-based methods and tools (Section 3.2.1) are key instruments for building driver scenarios of change in biodiversity and ecosystem services. Both approaches have their respective advantages, with participatory approaches facilitating multidisciplinary stakeholder participation and the inclusion of indigenous knowledge, while expert-based approaches allow for the greater use of formal modelling techniques and scientific knowledge. Different types of approaches and models are described in this chapter, which can be used (separately or together) at different scales and to describe specific changes in biodiversity and ecosystems, as well as their linkages.

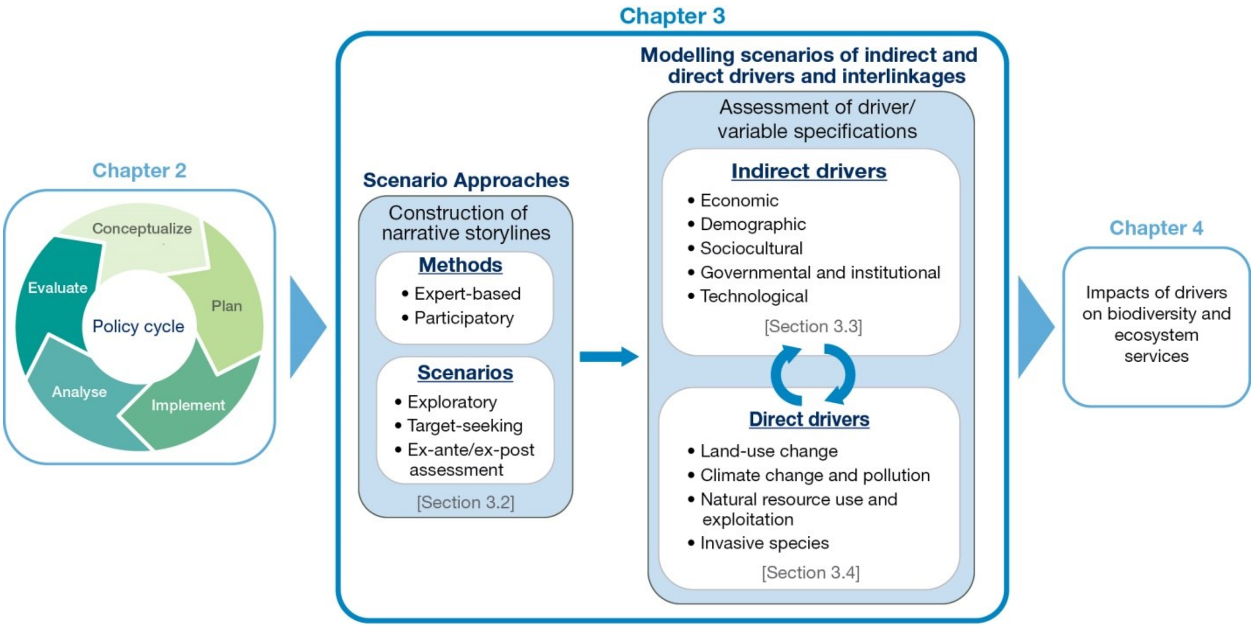



**Figure 3.1:** Chapter 3 overview**.**

Scenario construction (Section 3.2.2) begins with the development of qualitative storylines that are translated into driver scenarios. Modelling scenarios of indirect and direct drivers of biodiversity and ecosystem services (Sections 3.3 and 3.4) is multifaceted and in many cases multiple models are required to address multi-sectoral issues on different driver scales. The chapter then concludes with lessons learned and the way forward for future work on building scenarios and models of drivers of change in biodiversity and ecosystem services (Section 3.5).

# 3.2 Methodological approaches to scenario and model construction

The choice of method is crucial to the assessment of indirect and direct drivers. This choice depends strongly on the questions as well as the scope and scale of analysis. In this section, the different methodological approaches for assessing indirect and direct drivers in relation to the context of use are outlined. Many methods start with either expert-based or participatory techniques to identify relevant indirect drivers and construct scenarios. Based on the scenario assumptions, different types of modelling tools are used to quantify the evolvement of these indirect drivers and their impacts on the direct drivers.

## 3.2.1 Approaches

Expert-based approaches entail the use of expert opinion, knowledge (including scientific theory) or judgment to inform the various aspects of constructing scenarios and models of drivers. The term ‘expert’ implies an individual who has expertise or experience within a particular dimension through training, study or involvement in practice (Raymond et al., 2010). Participatory methods and tools help define complex problems related to the governance of drivers impacting particular biodiversity and ecosystem services. They also provide a platform for views to be aired, perspectives broadened, and a greater understanding of the policy issue under consideration. Including indigenous and local knowledge provides a more comprehensive reflection of prevailing conditions and other key inputs, and incorporates methods and approaches that capture holistic values that people place on nature while internalising principles and ethical values specific to their world views and realities (Illescas and Riqch'arina, 2007; Medina, 2014).

#### 3.2.1.1 Expert-based approaches

Although all scenario construction implicitly involves some degree of expert opinion, formal expert-based scenario modelling entails identifying and eliciting information from multiple experts, either individually or in a group (Krueger et al., 2012). To determine whether expert opinion should be utilised, Kuhnert et al., (2010) provided the following steps: 1) articulation of research questions, 2) assessment of available empirical data and whether the data can provide a solution, and if it can, 3) verification that sufficient resources are available to carry out the elicitation. Expert knowledge can also be utilised in studies where requisite sampling over spatial and temporal scales is not possible due to financial and/or logistical constraints (Martin et al., 2005).

Expert-based approaches are particularly valuable for translating a perceptual model (i.e. qualitative understanding) into a formal model (i.e. mathematical representation) (Krueger et al., 2012). In addition to the contributions to formal modelling, expert opinion can enter models through informal vectors such as subjective choices and value-laden assumptions (see Box 3.1), as well as other biases consistent with the experts' respective disciplinary training and background (Krueger et al., 2012).

Expert-based approaches are particularly susceptible to scientific uncertainties including subjective judgment and uncertainties associated with the parameterisation and weighting of variables.

Furthermore, the use of heuristics and the presence of cognitive bias associated with determining statistical probabilities can result in systematic bias throughout expert elicitations (Kynn, 2008). Disadvantages of expert-based approaches often include limited knowledge of local biota and ecological processes (Stave et al., 2007), which can significantly increase the time and resources needed to conduct environmental assessments. While the selection of, and disagreement among, experts can pose obstacles to this method of scenario construction (as well as the cost and time involved in eliciting information), scientists are increasingly aware of the advantages of the deliberate formal use of expert opinion to inform ecological models.

Experts can also be stakeholders – both experts and stakeholders vary in the degree to which they have expert knowledge as well as the extent to which they effectively have a stake in the issue under consideration (Krueger et al., 2012).

Experts can have significant institutional and financial interests, while scientific knowledge is not necessarily confined to traditional academic and research environments (Cross, 2003). The distinction between experts and stakeholders therefore needs to be undertaken carefully, with the understanding that experiential knowledge will impact the type of uncertainty introduced into the model, including individual bias. However, there are reliable techniques, such as the Delphi technique (see Box 3.1), that successfully reduce many uncertainties associated with expert-based elicitations.

**Box 3.1:** The **Delphi Technique**

Initially developed by the RAND Corporation in the 1950's, the Delphi Technique is a well-established method for eliciting the opinion of multiple experts – ideally between 10 and 18 (Okoli and Pawlowski, 2004) – used to construct scenarios and support decisions (Rauch, 1979). This method is particularly valuable in data-poor environments when translating qualitative responses into quantitative variables or subjective probabilities (Ouchi, 2004; MacMillan and Marshall, 2006) and is thus ideal for expert-based approaches to ecological modelling. The Delphi approach consists of consultations regarding the methodological approach, several rounds of independent and anonymous elicitation followed by feedback from experts leading to subsequent revisions and, resource-permitting, a workshop or meeting to address any remaining issues and crystallise final results. Under the guidance of an independent facilitator with knowledge in the field and experience in consensus-building, the controlled environment of the Delphi method promotes independent thought by preventing direct confrontation between experts (Dalkey and Helmer, 1963). This method has the benefit of reducing undue influence by individual members as well as mitigating the degree to which some members may be persuaded to conform (i.e. group think). Here, anonymity throughout the elicitation and revision cycles also serves to diminish other psychological bias inherent to group processes such as emergent group norms and gender-related process strategies (e.g. Haidt, 2001; Hannagan and Larimer, 2010).

#### 3.2.1.2 Participatory approaches

Participatory approaches to scenario development consist of involving a larger group of stakeholders through workshops or other formal meetings to share ideas and ultimately develop scenarios based on their collective knowledge.

This approach has the benefit of mobilising local and indigenous expertise on scenarios, as well as enabling participation and better informing local stakeholders (Patel et al., 2007; Palomo et al., 2011). Tools such as Fuzzy Set Theory assist in the co-production of knowledge between experts and stakeholders through the quantification of key scenario and model parameters (Kok et al., 2015). If properly conducted, participatory approaches help increase the effectiveness of environmental and biodiversity management (Palomo et al., 2011). Nonetheless, barriers to such approaches include the limited understanding of relevant issues – in particular the influence of exogenous drivers (those beyond the control of participants) and inter-scale (global, regional, national, local) interactions (MA, 2005a) – and considerable differences in opinion among participants as well as difficulty in translating qualitative data into quantitative inputs (Walz et al., 2007).

Among participatory approaches, the ‘agent-based participatory simulation’ method is a valuable way to investigate complex issues arising from natural resource management (Bousquet et al., 2002; Briot et al., 2007). Essentially, direct and indirect drivers of the depletion of biodiversity and ecosystem services are identified through a participatory exercise through a combination of role-playing games and multi-agent simulations. Relevant stakeholders are able to select the main indirect drivers and interactively construct numerous computer-based scenarios of collective governance for the improved conservation of biodiversity and ecosystem services. The combined multi-agent simulations/role-playing games approach has proven to be an effective means of establishing sustainable and inclusive management schemes for protected areas that are under pressure. The key advantage of such an approach consists of stimulating a participatory consultation process which fosters a sound collective effort to identify relevant indirect and direct drivers of the transformational process and to formulate scenarios and pathways of potential conservation and restoration of biodiversity and ecosystem services.

Stakeholder participation has, for example, proved critical when identifying drivers of change and their importance for an ecosystem approach to fisheries. Based on the Food and Agriculture Organization (FAO) code of conduct for responsible fisheries (Attwood et al., 2005) and the Australian ecological sustainable framework (Fletcher, 2002), a series of locally-adapted ecological risk assessments have been developed in the Benguela Current region (i.e. South Africa, Namibia and Angola) that take a participatory approach (Augustyn et al., 2014). This provides a transparent and structured process among stakeholders, which helps to prioritise the issues and drivers that need to be considered (Nel et al., 2007). Additionally, participatory approaches are frequently employed simply to map out a range of views among participants.

### 3.2.2 Scenarios

Scenario construction is a valuable endeavour when attempting to construct possible futures in the context of uncertainties, particularly when ecological outcomes are highly contingent on indirect drivers such as economic growth and demography (Carpenter, 2002).

Thus, scenarios or 'variants' are employed to account for uncertainty within models of the future. In these cases, rather than attempting to project from a specific set of values for driver variables onto a specific future, it is preferable to employ a variety of scenarios based on knowledge of a range of potential alternative futures (Peterson et al., 2003).

Exploratory scenario construction begins with the preparation of qualitative narrative storylines which provide the descriptive framework from which quantitative scenarios can be formulated. Such qualitative scenarios are particularly valuable as the temporal scale under examination increases and there are greater chances that exogenous influences may introduce unforeseen systemic change (e.g. a technological shift) (Rounsevell and Metzger, 2010). The use of qualitative scenario storylines and the subsequent parameterisation of key drivers has been well developed within the field of climate change research conducted by earlier IPCC assessments (Section 3.4.2). Here, the specification of model-based scenario assumptions has evolved considerably over time in response to scientific advances in our understanding of climate change as well as the acknowledgement that socio-economic drivers are an integral aspect of formulating potential futures (Abildtrup et al., 2006; Moss et al., 2010).

An extensive history of scenario building is beyond the scope of this paper (see for example Amer et al., 2013). Instead, an overview of scenario use within the decision-making context of the policy cycle, with a specific focus on exploratory and target-seeking scenarios as well as ex-ante and ex-post assessments, is provided (Table 3.1). Within this context, the choice of scenario and assessment type as well as the related methodological approach to scenario construction is highly contingent on the position in the policy cycle and the intended spatial scale.

**Table 3.1:** Combining scenario approaches and policy objectives.

| Approaches for using scenarios | Brief summary | Relevance for policy making processes | Role of indirect and direct drivers | Examples |
|---|---|---|---|---|
| EXPLORING alternative futures by using exploratory (descriptive) scenarios | Based on plausible alternative futures built on extrapolations of past trends and new assumptions | Creates awareness of future policy challenges and agenda setting. Assumes the **absence of explicit policy intervention** | Projections of indirect drivers and their effects on direct drivers | IPCC SRES, 2000; Global Environment Outlook (GEO)/UNEP; Millennium Ecosystem Assessment (MA), including from global to local applications |
| INTERVENTION: Using target-seeking scenarios (normative scenarios) | Starts with a prescriptive vision of the future and then works backward in time to visualise different pathways of achieving this future target | **Policy Prescriptive** Identifies the conditions necessary to achieve the desired target | Identification of driver values consistent with the desired target | IPCC Representative Concentration Pathways (RCPs) (Van Vuuren et al., 2011). VOLANTE European VISIONS on sustainable land use (Pedroli et al., 2015) |
| INTERVENTION: Policy screening using ex-ante assessment | Depicts the future effects of environmental policies | **Policy Screening** and impact assessment of alternative policy options before implementation | Driver projections are used as reference for policy options | The Strategic Environmental Assessment of the European Union (SEA Directive, 2001). Assessment of biofuel policies on direct and indirect land use change (e.g., Moser and Mußhoff, 2015) |
| POLICY EVALUATION using ex-post assessment | Looks backward to analyse the gap between environmental policy objectives and actual policy results, after using counterfactual scenarios | **Reactive Policy Assessment** Post hoc evaluation of policy effectiveness | Identification of drivers explaining discrepancies of outputs | For assessing forest loss within and outside protected areas (monitoring the success of protected areas) (Joppa and Pfaff, 2010) |

### 3.2.2.1 Exploratory scenarios

Exploratory scenarios (also known as 'descriptive scenarios') typically have both strong qualitative and quantitative components and are often combined with participatory approaches involving local and regional stakeholders (Kok et al., 2011). Exploratory scenarios frequently employ a co-evolutionary approach through the use of matrices where the projection of divergent futures is based on changes in the indirect and direct driver assumptions.

The relative benefits of exploratory scenarios include flexibility to construct storylines (conducive to greater creativity), coverage over a wide range of outcomes, and their application to problem areas where specific policy responses have yet to be formulated or the nature of the problem remains unclear (Van Vuuren et al., 2012a).

Exploratory scenarios are therefore particularly relevant in the agenda-setting stage of the policy cycle where the scale, relevant stakeholders and problem specificities are first addressed as the problem is brought to public attention (see Figure 3.2) (Stone et al., 2001). Exploratory scenarios can illuminate the discourse on the specific problems to be addressed by society in the presence of limited resources, by illustrating various potential futures starting from the current point in time.

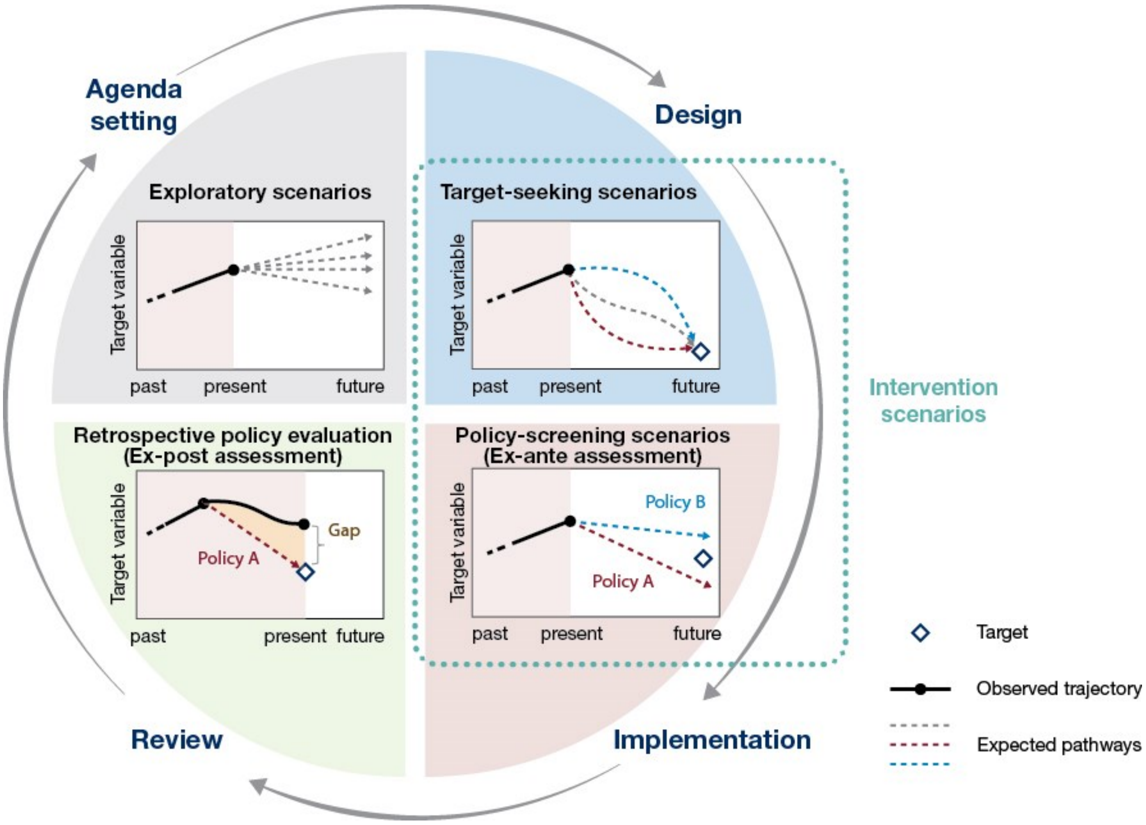



**Figure 3.2:** Building scenarios of indirect and direct drivers within the policy cycle context for biodiversity and ecosystem services.

Exploratory scenario approaches (see Box 3.2) have been utilised for climate change projections and were used in the IPCC assessments. This process started with the estimation of greenhouse gas (GHG) emissions as the major driver for climate forcing, leading to the Special Report on Emissions Scenarios (SRES) and the latest RCPs. These scenarios were initially applied at a global scale with regional scale scenarios typically constructed through downscaling (downscaling refers to the transformation of information from coarser to finer spatial scales through statistical modelling or the spatially nested linkage of structural models). Exploratory scenarios describe the future according to known processes of change or as extrapolations of past trends (IPCC, 2001).

In the absence of policy change, ‘business-as-usual’ or baseline scenarios represent a future with no major interventions or paradigm shifts in the functioning of a system.

However, the term ‘business-as-usual’ may be misleading in the policymaking process because exploratory scenarios can also describe futures that bifurcate at some point (e.g. due to the adoption or rejection of a new technology) or that make some assumptions about the functioning of a system. Exploratory scenarios are common in environmental studies because they require less speculation about the future and tend to be more ‘value-free’ compared with target-seeking or normative scenarios (Alcamo, 2001). Furthermore, researchers and stakeholders may be more comfortable with the forward progression of time in exploratory scenarios than with the backward-looking perspective adopted in target-seeking scenarios.

**Box 3.2:** Examples of exploratory scenario narratives for coral reef ecosystems in the Caribbean

***Main steps for building exploratory scenarios:***

1) Identification of research areas (regarding potential changes in biodiversity and ecosystem areas): global, regional, national or local (e.g. coral reef ecosystems in the Caribbean)

2) Identification of potential changes in biodiversity and ecosystems (e.g. increasing coral bleaching and mortality)

3) Identification of main drivers of change (direct and/or indirect drivers), for example: a) climate change (ocean acidification, higher temperatures, etc.), b) unsustainable socio-economic activities (tourism, fishing, etc.)

4) Selection of scenario axes and scenario logic (this example includes two axes to simplify the illustration for didactic purposes. In practice, several key stressors can generate pressures on biodiversity and ecosystems in a specific area):

- Climate change trends
- Socio-economic stressors in the Caribbean, particularly regarding unsustainable activities in coastal areas and oceans

5) Building preliminary scenarios:

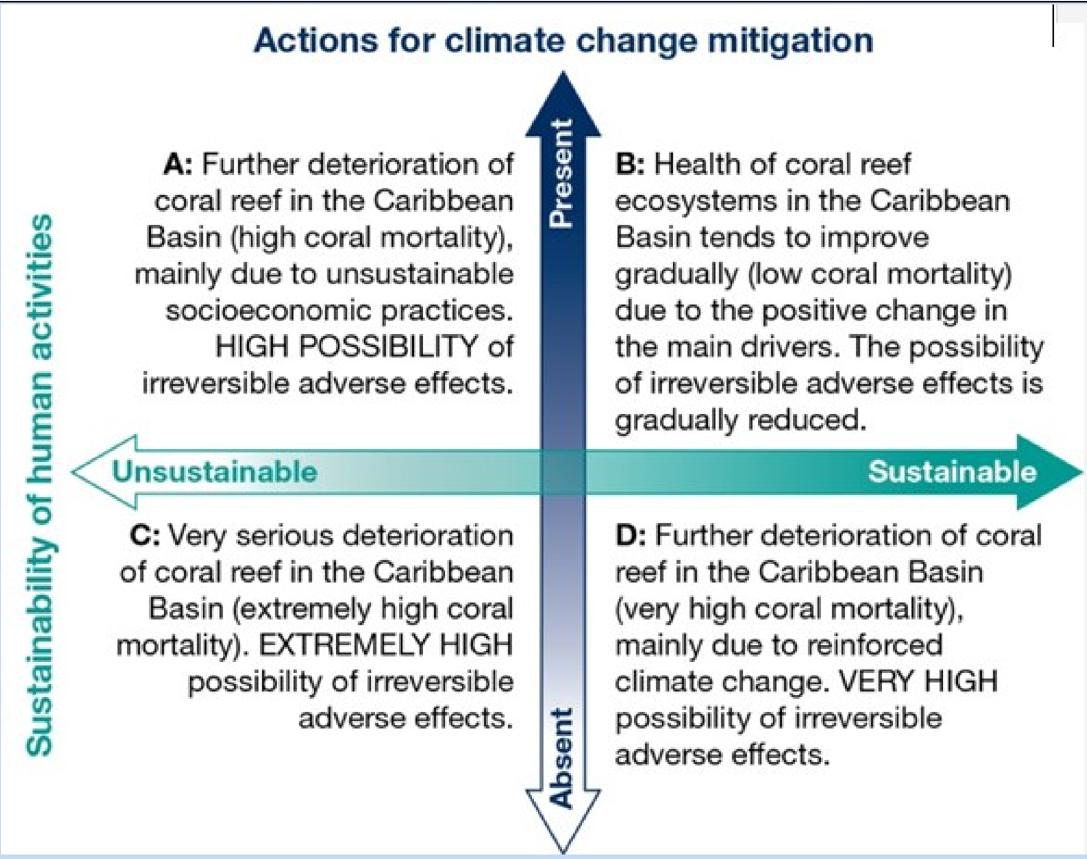



**Figure Box 3.2:** Scenario matrix

### 3.2.2.2 Target-seeking scenarios

Policy design, or formulation, is the stage in which the descriptive is transformed into the prescriptive according to the desired normative approach (Loorbach, 2010). Here, the will to address a recognised problem is translated into a viable policy formulation with clearly-defined objectives. For successful policies to be designed, policy options must be feasible in terms of economic and political resources as well as meet the needs of both the underlying science and interested stakeholders (Lemos and Morehouse, 2005; Jann and Wegrich, 2007). Employing normative pathway analyses such as backcasting approaches at this stage of the policy cycle allows for the identification of multiple potential pathways to a desired future vision. Target-seeking scenarios (also known as ‘normative scenarios’) constitute one subclass of the more general class of intervention scenarios (also known as ‘policy scenarios’) introduced in Chapter 1.

Target-seeking scenarios are a valuable tool for examining the viability and effectiveness of alternative pathways to a desired outcome, particularly when used in conjunction with appropriate decision-support protocols and tools such as those described in Chapter 2.

Target-seeking scenarios start with the definition of a clear objective or a set of objectives that can either be specified in terms of achievable targets (e.g. in terms of the extent of natural habitats remaining, or of food production self-sufficiency) or as an objective function to be optimised (e.g. minimal biodiversity loss).

Together with these goals and objective functions, a set of constraints is defined (e.g. excluding areas for conversion) to ensure realistic feasible outcomes. Backcasting (see Chapter 2) is particularly valuable when there is a great deal of uncertainty regarding future developments and the most likely future is not necessarily the most desirable (Robinson, 2003). Intervention scenarios typically encompass both the design and implementation phases (see Figure 3.2). Within this assessment, however, target-seeking scenarios and the subsequent ex-ante assessments (Section 3.2.2.3) are distinguished to highlight their relative contributions to weighing the relative desirability of different pathways.

**Box 3.3:** Example of target-seeking scenarios: zonation tools (Moilanen et al., 2009) for protected area allocation under the Aichi biodiversity target

According to Aichi biodiversity target 11 adopted by the Convention on Biological Diversity, the protected area network should be expanded to at least 17% of the terrestrial world by 2020. However, there is a considerable risk of ineffective outcomes due to land-use change and uncoordinated actions between countries. Recent research that used zonation tools to identify the optimum location of protected areas for biodiversity conservation shows that, with a coordinated global protected area network expansion to 17% of terrestrial land, the average protection of species ranges and ecoregions could triple (Pouzols et al., 2014). If projected land-use change by 2040 takes place, it becomes infeasible to reach the currently possible protection levels, and over 1,000 threatened species would lose more than 50% of their present effective ranges worldwide. In addition, a major efficiency gap is found between national and global conservation priorities. Strong evidence is shown that further biodiversity loss is unavoidable unless international action is quickly taken to balance land use and biodiversity conservation.

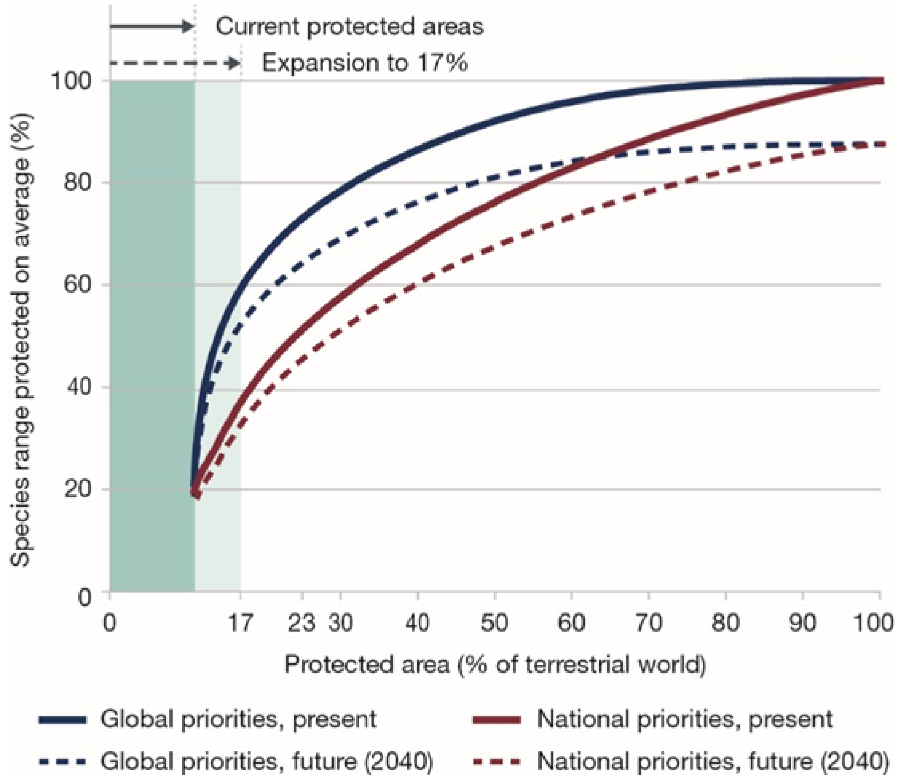



**Figure Box 3.3:** The relation between the protected area and the maximum attainable protection of species under conditions of the optimum spatial allocation of protected areas. Under global priorities the allocation is globally optimised, while under national priorities the optimisation is based on a country-by-country basis. Future conditions refer to conditions under the projected land-use change, which constrains the spatial allocation of protected areas (Modified by permission from Macmillan Publishers Ltd: [Nature] Pouzols et al., 2014, 516, 383–386, copyright 2014).

### 3.2.2.3 Ex-ante/ex-post assessment

Ex-ante and ex-post assessments of environmental policies are tools in the policymaking process. Ex-ante assessment is a proactive approach, oriented to identify and address potential effects of environmental policies. Many of the decision-support protocols and tools described in Chapter 2 provide a structured means of undertaking ex-ante assessments. This form of assessment typically makes strong use of a second subclass of intervention scenarios (introduced in Chapter 1).

Ex-ante assessments use policy-screening scenarios to forecast the effects of alternative policy or management options (interventions) on environmental outcomes.

Environmental Impact Assessment (introduced in Chapter 2) is a widely used tool within this perspective. Ex-ante assessment usually starts in the very early stages of a policy formulation and design. It may therefore contribute to the social acceptance of policies by anticipating and addressing conflicting objectives and adverse effects. When properly organised, this assessment may include expert considerations and consultations to relevant stakeholders such as government authorities, community representatives, non-governmental organisations and the general public. This assessment perspective is embodied, for instance, in the Strategic Environmental Assessment of the European Union (European Commission, 2001).

Other types of scenarios (e.g. target-seeking scenarios) can be used to complement and support ex-ante assessments. In some cases, these assessments are carried out through multiple scenario comparisons, and this approach helps policymakers compare the potential consequences of various scenario-based options (e.g. Helming et al., 2011). In the intervention design phase, different alternative policy options or management strategies are often developed. While final decisions will be heavily influenced by the full political and societal context, scenarios and models can better inform such decisions by investigating the effectiveness and unintended consequences of proposed policy measures through ex-ante assessment (Helming et al., 2011). Policy-screening scenarios require a detailed specification of changes in drivers such as uptake of policy measures on human behaviour, often focusing on shorter, more policy-relevant time frames than other types of scenarios. Economic and sector-based models are especially dominant here as the economic consequences and cost-benefit assessment of the proposed changes in drivers are essential in decision making.

The policy review phase involves the ex-post reflective assessment of the extent to which the policy implementation achieved the goals outlined in the initial stage of problem identification. In practice, evaluations are rarely consistent with underlying theory which stipulates that multiple criteria and methods are used, formal policy goals are questioned, and stakeholders are actively involved throughout the process (Mickwitz, 2003; Huitema et al., 2011).

Ex-post assessments are the present evaluations of past efforts to achieve policy goals throughout all stages of the policy cycle and decision-making context.

Some key obstacles to the realisation of policy goals include instrument design oversight, inadequate monitoring, and an absence of effective enforcement mechanisms (Haug et al., 2010). Furthermore, due to the inherent complexity of the environment-policy nexus, the enactment of environmental policies may result in impacts that run counter to the original goals or encourage counterproductive behaviour such as rebound effects (Faber and Frenken, 2009).

Ex-post assessments can be based on the straightforward monitoring of variables of interest as well as on a comparison of the achieved change or status with the original targets and the anticipated impacts of the implemented measures. In many cases, it is important to distinguish the effects of the implemented policy or management scheme from autonomous developments (Hoffmann et al., 2015).

Econometric models are used to evaluate the contribution of different conditions to the monitored data. For example, straightforward ex-post assessments may assess forest loss within and outside protected areas to monitor the success of protected areas. However, such straightforward evaluations may be biased by the different locations of protected and unprotected natural areas that heavily impact the risk of deforestation (Joppa and Pfaff, 2010a). Under such conditions, more sophisticated techniques for ex-post assessment need to be applied that are able to distinguish the influence of such confounding factors on the monitored impacts.

## 3.2.3 Models

Many typologies of modelling tools of indirect and direct drivers and their interactions are possible. Modelling tools can for example be categorised depending on their qualitative or quantitative nature, whether the underlying phenomenon can be represented by structural equations or driver processes are captured by data-driven approaches, and whether the model is of a deterministic or stochastic nature. Such broad typologies can typically be further broken down into sub-categories. For example, a distinction is made among structural models between simulation models and normative target-seeking models. Among the latter, classical economic models typically maximise a welfare function or minimise costs. If such models cover the entire economy they are referred to as general equilibrium models, while partial equilibrium models cover a specific sector in greater detail. Such economic models can be constructed for comparative static analysis to analyse the introduction of new drivers such as policy shocks or for dynamic assessments to analyse solution pathways.

### 3.2.3.1 Modelling methods

Traditionally, structural economic models simulate indirect and direct drivers in deterministic settings and the latest developments in these models allow for the assessment of very uncertain and stochastic phenomena such as the impact of climate change (Leclère et al., 2014) or agricultural production volatility on land-use change (Fuss et al., 2015). Short-term forecasts of drivers, most frequently economic drivers, are generated by non-structural models, implying that the modelling tool finds patterns in the data itself and projects these into the future. Tools for the extrapolation of current trends include statistical and econometric methods and data mining tools such as artificial neural networks, rough and fuzzy set approaches, and network theory approaches. These tools also allow for projections of an ensemble of variables that interact with each other, such as vector autoregressive models.

Data-driven models will not typically allow for a mechanistic understanding of how and why drivers interact. As a general rule, the short-term predictive skill of data-driven approaches is superior to mechanistic structural models. However, for long-term analyses – where biophysical boundaries of production systems need to be respected – and for the analysis of structural adjustments of drivers due to policy changes, mechanistic models are more suitable.

**Good modelling practice**

Modelling of indirect and direct drivers of change in biodiversity and ecosystem services has so far been undertaken mainly in the domain of academic research and thus good modelling practice is defined through the peer review process.

Key driver scenarios such as long-term Gross Domestic Product (GDP) development are produced through more expert-driven simple models and are not subject to stringent technical quality control measures; therefore the credibility of such driver projections typically rests on the reputation of the expert team.

There are currently less than a handful of institutions that issue long-term projections of GDP, and none of their models consider feedback from resource constraints. More sectoral models of indirect drivers, such as integrated assessment models or partial equilibrium models, are typically very large and highly complex due to their fundamentally non-linear structures. It is next to impossible to review such model structures with reasonably limited resources; if operated by an individual, analyses generated by such models are typically judged on the behaviour of a few output variables of interest given a specific problem. Integrated assessment models are typically used at the stage of policy formulation and very few of these models are actually used for policy planning purposes where review procedures are more biting than academic peer review. Given the fact that there are fundamentally different purposes and subsequent review procedures for different modelling tools, the production of consistent scenarios of long-term driver behaviour is currently more an art than a science. It is unlikely that there will be a major breakthrough in the science of long-term projections of indirect and direct drivers. Rather, there is a tendency to increasingly introduce quality control measures through good practice guidance.

For example, good practice guidance for GHG accounting in the land-use sector has been established for more than a decade, and this provides the basic accounting rules for subsequent projections. The modelling process of producing projections is subject to TCCCA principles (transparency, completeness, consistency, comparability and accuracy). For example, in establishing forest management reference level (FMRL) scenarios, the TCCCA principles allow a technical evaluation of these scenarios by an independent review panel organised by the United Nations Framework Convention on Climate Change (UNFCCC). The ultimate purpose of the FMRL process is to trigger payment streams for additional climate mitigation efforts.

#### 3.2.3.2 Linking multiple models

The development and quantification of scenarios of indirect drivers and their impacts on direct drivers of change in biodiversity and ecosystem services is multifaceted. In many cases, multiple models are required to operate at different spatial scales and/or to cover various driver constellations. For example, modelling of habitat conversion may require the use of demographic, economic and biophysical models to properly represent the development of the impact of different indirect drivers. For regional assessments, global scale assessment models are often required to account for the influence of distant drivers on the region of interest, while region-specific models are used to add finer spatial detail to the simulations (Verburg et al., 2008).

No single model can capture all dynamics at a high level of detail, and the coupling or integration of models has become a popular tool to integrate the different dimensions. However, the degree of coupling varies among studies and the choice of integrated modelling versus a loose coupling of models depends on the specific requirements of the assessment as well as the system under consideration.

The loose coupling of specialised models has the advantage that the specific strengths of each model are retained. An example of this tactic is the nested modelling approach used by Verburg et al., , (2008). Here, global economic models explore changes in world consumption and production in terms of the consequences for land use at the level of world regions. Detailed, spatially-explicit land-use change models subsequently downscale calculated areas of land use to individual pixels to show the types and location of changes in land use and terrestrial habitats. Based on the resulting land-use change patterns, a new set of models is used to assess the consequences of land-use change for carbon sequestration (Schulp et al., 2008) and ecosystem services.

The disadvantage of loose coupling models where only limited information is exchanged between the models (often in only one direction) is the lack of representation of feedback between the modelled components and the risk of inconsistencies in representation of the same phenomenon in the different models (e.g. a forest in one model can be defined differently in another model).

The loose coupling approach has a risk of propagation of error and uncertainty between the coupled models, which is difficult to track and quantify (Verburg et al., 2013b).

At the other end of the spectrum, integrated assessment models have been developed that embed the different model representations of the system in a consistent manner. Often, such integrated assessment models are modular and the different modules are built based on simple representations of the system under consideration. Given the embedding in a single simulation environment, the inclusion of feedback and interaction between the different modules is allotted more attention and there is consistent representation of variables across the different modules (Verburg et al., 2015).

Similar models have been developed for regional scales that include the most important spatially-specific indirect and direct drivers while taking into account knowledge on region-specific interactions and data availability (Harrison et al., 2015). A disadvantage of this approach is the inherent complexity of the models and the strongly simplified representation of the individual model components. This increased complexity reduces the applicability and transparency of the models (Voinov and Shugart, 2013). Although presently these models tend to be used for a wide range of different questions, their model structures often inherit a focus on the specific questions that the models were developed for. Therefore, care needs to be taken regarding the range of their application.

The choice of integrated modelling versus a loose coupling of models depends on the specific requirements of the assessment but also on the system being studied. An integrated modelling approach is required when feedback between the system components or spatial scales studied is important to system outcomes. However, when dynamics in the individual components dominate, the use of specialised models is recommended to capture such dynamics adequately. Also, should the study aim to identify leverage points in the dynamics of the indirect drivers, a loosely coupled model approach may have advantages for studying the different components of the system both separately and as part of the full system, allowing identification of the role of system interactions.

## 3.3 Scenarios and models of indirect drivers

The role of indirect drivers is an integral aspect of scenario development and subsequent analysis in complex ecological systems. Indirect drivers play a major role in influencing direct drivers of biodiversity and ecosystem change, as well as strongly influencing other indirect drivers. Socio-economic and demographic trends heavily influence consumption patterns with subsequent environmental implications (e.g. Seto and Kaufmann, 2003). In addition to interacting with socio-economic and demographic drivers, technological innovation can lead to the adoption of cleaner and more sustainable energy production, as well as indirectly contributing to environmental degradation through electronic and other waste as well as increased demand for the raw materials used in new technologies. While difficult to model, an understanding of the role of societal drivers such as culture and government is crucial to sustainable ecosystem management as these are strong drivers of value sets and decision frameworks that affect behaviours.

The influence of indirect drivers on biodiversity and ecosystem change materialises to a large extent through the valuation of biodiversity and ecosystem services. Institutional setups, as well as environmental policies and governance frameworks, are currently embedded in shaping valuation outcomes, with long-term effects for biodiversity conservation and equity of access to ecosystem services benefits (Gomez-Baggethun and Ruiz-Perez, 2011). Elaborated upon in subsequent sections, the relative levels of different types of uncertainty (defined in Chapter 1) and the extent of the current use of indirect drivers in scenarios and models varies from driver to driver (Table 3.2).

**Table 3.2:** Degree of uncertainty and utilisation in scenarios and models by indirect driver.

| Drivers | Utilisation in scenarios and models | Stochastic uncertainty | Scientific uncertainty | Linguistic uncertainty |
|---|---|---|---|---|
| Economic | High | Low | Low | Low |
| Demographic | High | Low | Low | Low |
| Sociocultural | Low | Low | High | High |
| Governance and institutions | Medium | Medium | Medium | Medium |
| Technological | High | High | High | Low |

### 3.3.1 Economic trends

Economic drivers and economic trends impact both social and environmental dimensions of sustainable development. Economic growth is the main global driver of resource consumption (Dietz et al., 2007). Consequently, these drivers have a growing effect on ecosystems and ecosystem functions (Gomez-Baggethun and Ruiz-Perez, 2011). According to the MA (MA, 2005c), global economic activity increased nearly sevenfold between 1950 and 2000 and is expected to grow again by a further threefold to sixfold as measured by GDP by 2050. While technological and institutional innovations have increased resource-use efficiency, consumption growth has outstripped increases in efficiency (Raudsepp-Hearne et al., 2010).

Taking a historical perspective, past and prevailing patterns of production and consumption embodied in global economic trends have generated growing pressures on natural resources, the environment and ecosystem functions. The World Wildlife Fund Living Planet Report (McLellan et al., 2014) concludes that humanity's demand has exceeded the planet's biocapacity for more than 40 years, and the ecological footprint shows that 1.5 Earths would be required to meet the demands humanity makes on nature each year. This demand is further compounded by the influence of population trends (see Section 3.3.2) and technological change (see Section 3.3.5).

GDP is widely used as the sole socio-economic measure. Alternatively, the Human Development Index (HDI) adopts a wider approach, taking into account quality of life, health and education (see UNDP, 2014a). However, even the HDI has considered the economic component (income) as a key factor in its calculations since 1990, when the publication of the annual United Nations Development Programme (UNDP) Human Development Report started (UNDP, 2014b). Virtually all socio-economic and environmental scenarios for this century (i.e. up to the year 2050 and beyond) include economic growth as a key driver, and GDP scenarios are typically built on explicit storylines about the evolution of determinants of the economic system.

For example, the identification of possible elements of SSP scenarios (O'Neill et al., 2014) consider the following scenario elements essential within the category of 'economic development': global and regional GDP, or trends in productivity; regional, national and subnational distribution of GDP, including economic catch-up by developing countries; sectoral structure of national economies, in particular the share of agriculture, and agricultural land productivity; share of population in extreme poverty; and nature of international trade. More information on the SSPs, including economic and demographic projections, can be found in the SSP database (https://tntcat.iiasa.ac.at/SspDb).

According to the IPCC Fifth Assessment Report (IPCC, 2014), economic and population growth continue to be the most important indirect drivers of $CO_2$ emissions. This assessment highlights that the contribution of population growth between 2000 and 2010 remained roughly identical to the previous three decades, while the contribution of economic growth rose sharply.

Scenarios that assume rapid economic growth in the coming decades are mainly based on prioritising market goals and incentives under conventional market approaches, with adverse social and environmental implications, including negative impacts on biodiversity and ecosystems (e.g. Global Environmental Outlook 4 (GEO4) Market First, Rothman et al., 2007) (IEEP et al., 2009).

The linkages between economic drivers and technological development have also been explored in the context of building socio-economic and environmental scenarios. In many cases, scenarios assuming rapid economic growth in a conventional market context are based on dynamic technological development. However, many multidimensional asymmetries characterise these processes.

### 3.3.2 Demographic trends

In concert with other indirect drivers, changes in population size as well as demographic variables such as population distribution and age structure exert significant anthropomorphic pressures on direct drivers of biodiversity and ecosystem change. Demographic pressures are intricately interlinked with consumption and environmental externalities, many of which exhibit non-linear dynamics not regulated by market forces (Dasgupta and Ehrlich, 2013). In addition to greater demand for natural resources, growing populations require greater amounts of food, driving land-use and land-cover change through deforestation and conversion to agricultural land. Populations with high per capita consumption rates (of goods and services) generate high demand for natural resources, representing a potentially greater biodiversity and ecosystem services threat than population growth (see Section 3.3.3).

Urbanisation driven by growing populations and internal migration acts as an indirect driver of land-use change through linear infrastructures such as transportation networks and synergies with other forms of infrastructure development (Seiler, 2001).

In addition, while the effect of urbanisation on local land-use change is a complex phenomenon contingent on a number of factors, outmigration to urban areas frequently results in greater mechanisation and agricultural intensification made possible by remittances and driven by higher urban consumption levels (Lambin and Meyfroidt, 2011).

The primary determinants of population growth and structure are fertility, mortality and migration, with fluctuations among the former two characteristic of stages in the demographic transition model (e.g. Caldwell et al., 2006). Regional and local variation exists where there are significant socio-economic, governmental and developmental heterogeneities, particularly between rural and urban areas of less developed countries. The most recent United Nations (UN) population projections (UN, 2015) utilise Bayesian hierarchical models and the cohort component method to formulate probabilistic forecasts of population growth, adding to the high/low/medium scenarios of past UN projections (Gerland et al., 2014). Whereas the UN projects continued growth throughout this century, the International Institute of Applied Systems Analysis (IIASA) projects an 85% chance of global population stabilisation and relies more heavily on expert-based assumptions, utilising a multi-state cohort model to produce projections by age and sex, differentiated by education (Lutz et al., 2014). Here, projections are formulated according to five SSPs and contingent assumptions for fertility, mortality, migration and education.

While the focus in the field of demography is on global and national population projections, future research is increasingly taking into consideration subnational migration patterns and differential population trajectories according to socio-economic heterogeneities. Such analyses will be of considerable importance for understanding the effect of population growth on biodiversity and ecosystem change at regional and local spatial scales. As one example, population age structure has been found to influence consumption patterns, with younger and older people consuming more than middle-aged cohorts (e.g. Erlandsen and Nymoen, 2008; Liddle and Lung, 2010). This illustrates the paramount importance of examining how people interact with their environment due to socio-economic (Section 3.3.1) and sociocultural (Section 3.3.3) influences.

### 3.3.3 Society and culture

Culture in the form of the values, norms and beliefs of a group of people can act as an indirect driver of ecosystem change by affecting environmentally-relevant attitudes and behaviours. Chapter 5 provides an elaboration on the role of values (see also IPBES Deliverable 3d *on the conceptualization of multiple values*). The influence of societal and cultural values (and subsequent behaviour) on indirect and direct drivers of biodiversity and ecosystem change is acknowledged throughout the existing literature (e.g. Milton, 2013).

The impact of sociocultural influences on drivers of biodiversity and ecosystem change is often not explicitly captured in formal modelling methods due to the difficulty of identifying and parameterising what are often complex and overlapping phenomena.

In this respect, the role of sociocultural heterogeneity is frequently overshadowed in modelling applications by more easily quantified socio-economic metrics (e.g. GDP and education), prompting criticism that data-driven methodologies place an undue emphasis on measurable indicators while neglecting the role of sociocultural values and practices.

In addition to the challenge of identifying and measuring sociocultural drivers that capture the way in which people interact with their environment, understanding environmentally-relevant attitudes and values is further complicated by the value-action gap (Blake, 1999; Kollmuss and Agyeman, 2002). There is a large body of quantitative research from the cognitive sciences highlighting the considerable disparity between knowledge, values and actual behaviour, indicating that rationalist linear models do not fully capture the processes underpinning decision-making behaviour (e.g. Bechara et al., 1997; Haidt, 2001). Research into social networks reveals that behaviour is substantially shaped by the sociocultural context in which individuals are embedded (Christakis and Fowler, 2013). These dynamics also apply to pro-environmental behaviours with, for example, the use of block leaders to disperse information on conservation through community and social networks (Abrahamse and Steg, 2013). The growing field of social network analysis thus represents one statistically rigorous method of identifying individuals who are the most influential in spreading information and values through their respective peer networks (i.e. high centrality individuals) (Burt et al., 2013).

Due in part to their highly interlinked and amorphous character, sociocultural values are greatly affected by other indirect drivers. For example, in India researchers have largely attributed low meat consumption to cultural and religious traditions that prohibit and discourage the consumption of meat, particularly beef (Godfray et al., 2010b). Although India is known as one of the world's most vegetarian-friendly countries, a closer examination reveals a considerable amount of heterogeneity in India's diet and a trend toward the adoption of Western consumption patterns (Amarasinghe et al., 2007; Deaton and Drèze, 2009). Livestock production has a substantial negative impact on biodiversity through a number of direct drivers, including meat production-related habitat loss, indirect and direct GHG emissions, land degradation caused by excessive grazing and nutrient pollution (Stehfest et al., 2009; Machovina et al., 2015). Due to the considerable environmental impact of meat-heavy diets (Herrero et al., 2013), scenario analyses often include meat, vegetarian and healthy diet variants (e.g. Stehfest et al., 2009; Wirsenius et al., 2010).

### 3.3.4 Governance and institutions

Institutions play an important role in the management and exploitation of biodiversity and ecosystem services (Lowry et al., 2005; Abunge et al., 2013). Ill-informed and weak governance frequently leads to mismanagement of the commons (see Box 3.3), as well as the adoption of environmentally-unsustainable policies (Laurance, 2004; UNEP, 2013). Effective institutional design and implementation is however crucial. Institutional drivers operate at various spatial scales, from global (international) to local (subnational), and include the influence of policies that encourage a particular behaviour (e.g. agricultural subsidies) as well as the direct impact of enacting environmental legislation (e.g. designation of conservation areas). The concept of governance used by Gupta and Pahl-Wostl, (2013) refers to the exercise of authority by different social actors through the development and implementation of explicit and implicit substantive and procedural rules to manage resources for the social good.

In many countries, factors such as weak governance and institutions, lack of cross-sectoral coordination and illegal activity are cited as key indirect drivers of ecosystem change (Kissinger and Rees, 2010). Common governance challenges include confused goals, conflicts and unrealistic attempts to scale up beyond institutional capacity. Where collective action and conflict resolution mechanisms break down, the governance of ecosystem resources is compromised (Ostrom, 1990). Fragmented legal systems can lead to gaps and conflicts (Techera and Klein, 2011, Pomeroy et al., 2010), while the governance of large-scale ecosystems requires the 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 the exposure of coastal communities to catastrophic storm and tsunami events (Barbier et al., 2011). Alternatively, public policies can positively affect biodiversity and ecosystem services dynamics as exemplified by recovering fish stocks under the Common Fisheries Policies of the European Union (Fernandes and Cook, 2013). Here, secure private-property rights are widely considered to promote more efficient resource utilisation and property management than open access schemes, although there are many circumstances in which private-property rights do not guarantee resource conservation (Acheson, 2006), in addition to which most common property arrangements involve some degree of private-property management (Ostrom and Hess, 2007). Group size and makeup (e.g. gender) also have important implications for sustainability in situations involving collective resource management (Poteete and Ostrom, 2004; Westermann et al., 2005).

Governmental and institutional norms condoning corruption can easily become entrenched in impoverished environments, with significant consequences for the sustainable management of biodiversity and ecosystem services.

The failure to enforce rules (e.g. due to corruption or underfunding), as well as the absence of clear boundaries at the local level, can lead to collective action problems (Gibson et al., 2005).

So-called 'paper parks' are one example of where intended conservation measures lack the political willpower or enforcement capabilities necessary to carry them out (Wright et al., 2007). The problem of corruption is particularly pronounced when the enforcement of rules regarding highly-valued resources hinges on the ability of poorly paid government officials to resist bribes (Smith et al., 2003). Furthermore, the sustained impacts of direct drivers such as natural disasters can result in governmental and institutional instability, highlighting potential feedbacks between indirect and direct drivers (see Box 3.4).

**Box 3.4:** Divergent environmental management histories in Haiti and the Dominican Republic

The effects of institutional and governmental policies on the environment is clear in the contrast observed between the Dominican Republic and Haiti. Despite geographical similarities, a long history of weak environmental governance coupled with colonial exploitation has led to ecosystem degradation and increased vulnerability to natural disasters in Haiti (Roc, 2008). In addition to biodiversity protection and preservation, forest conservation measures as well as planning and adaptation capacities are crucial aspects for reducing the impact of natural disasters on human life and development (Day, 2009). In contrast with Haiti, the Dominican Republic has largely mitigated such consequences through successful environmental management. Where Haiti's forested territory has shrunk from approximately 85% in the 15$^{th}$ century to 2–4% today, forest cover in the Dominican Republic has rebounded from 12% in the 1980s to 40% today, due in large part to reforestation programmes and the enforcement of regulations. In Haiti, land degradation resulting from deforestation and unsustainable agricultural practices is a major direct driver of ecosystem change, with trade in charcoal providing a strong economic impetus. In contrast with the constitution of the Dominican Republic, which prioritises sustainable environmental management, many of the relevant laws in Haiti date back to the 19$^{th}$ century and the enforcement of extant regulations is hampered by a lack of political will as well as technical and financial limitations.

International trade and financial policies and practices considerably influence biodiversity and ecosystems services. Trade liberalisation, for instance, may have positive impacts to the extent that it stimulates the more efficient use of resources on macro-scales and connects more regions to the world market. However, higher levels of foreign debt service, structural adjustment programmes and a high dependency on primary sector exports are associated with higher numbers of threatened mammals and birds. This is because structural adjustment loans and large debt service burdens lead debtor nations to increase exports of agricultural goods and natural resources to generate currency for debt repayment (Shandra et al., 2010). Finally, conflicts undercut or destroy environmental, physical, human and social capital, diminishing available opportunities for sustainable development (UNEP, 2006).

The vital role of governance and institutions as drivers of biodiversity and ecosystem change was highlighted in the ALARM project, with scenarios encompassing agricultural, chemical, energy, transport, technology and trade sector policy variants (Spangenberg, 2007). The future application of the current ecosystem services approach will need to involve a more critical focus on environmental governance, transparency and participation as well as a consideration of the great uncertainties prevailing at various spatial and temporal scales (Paavola and Hubacek, 2013).

A more thorough understanding of how biodiversity, ecosystems and ecosystem services are governed, and incorporation of this understanding into driver scenarios, will be crucial for ensuring improved biodiversity and ecosystem services management in the context of governance systems.

### 3.3.5 Technology

The rate of technological change is considered to be an indirect driver of biodiversity and ecosystem services change because it affects the efficiency with which ecosystem services are produced or used (Alcamo et al., 2005). It is recognised that technological change can result in increased pressure on ecosystem services through increased resource demand, as well as leading to unforeseen ecological risks. In comparison with anthropomorphic indirect drivers that are relatively constrained by biophysical limitations such as economic and demographic trends, technological innovation can potentially serve as a catalyst of paradigmatic shifts in production systems with considerable societal implications (e.g. Perez, 2004). Although technology can significantly increase the availability of some ecosystem services and improve the efficiency of the provision, management and allocation of different ecosystem services, it cannot serve as a substitute for all ecosystem services (Carpenter et al., 2006).

The impact of technological innovation on biodiversity and ecosystem change is exerted through its influence on direct drivers as well as through interactions and synergies with other indirect drivers. With the exception of recent work (e.g. Dietrich et al., 2014), the role of technology trends in land-use change modelling applications is typically implemented exogenously due to the relative paucity of information on the relationship between research and development and technological change. Such decoupling of the assumptions about technological change from model dynamics can result in an underestimation (or, potentially, overestimation) of technological change that is most problematic in long-term projections (Dietrich et al., 2014). As with economic and demographic drivers, scenarios of technological change are included in the SSPs.

Technologies associated with agriculture and other land uses (see Box 3.4) have a large impact on drivers of biodiversity and ecosystem change. The agricultural intensification of the 'green' revolution led to higher crop yields and lower food prices, to some extent mitigating the expansion of agricultural land (Evenson and Gollin, 2003) and resulting in a net decrease in GHG emissions (Burney et al., 2010). However, while intensification may have represented an advantageous pathway from a land-use change and climate change perspective, excessive nitrogen and phosphorous use through fertilisers has led to the substantial degradation of freshwater and marine habitats (Smith et al., 1999). Furthermore, the shift from traditional crop varieties to industrial monocultures has resulted in a loss of crop genetic diversity (FAO, 2010) as well as increased susceptibility to disease and pests (Zhu et al., 2000; Jump et al., 2009). Looking to the future, recent global food demand projections foresee a doubling of crop production between 2005 and 2050 (Tilman et al., 2011), largely due to the global dietary shift toward greater rates of meat consumption now taking place throughout the developing world (Delgado, 2003; Speedy, 2003; Thow and Hawkes, 2009).

Agricultural land expansion is estimated to be the direct driver for around 80% of deforestation worldwide and is the dominant cause of land-use change (Hosonuma et al., 2012) as well as a key contributor to GHG emissions through land-use change (Paustian et al., 2006).

Agricultural technologies acting on direct drivers of biodiversity and ecosystem change include improvements in crop yields and resilience; sustainable livestock, fishing and aquaculture practices; and mechanisation and engineering practices such as precision farming (Beddington, 2010). In addition to shaping current practice, the introduction of new technologies can result in entirely new markets, particularly in confluence with government incentives, as illustrated in the case of biofuels (see Box 3.5). In a potential future of nine billion inhabitants, some argue that genetically modified crops hold the promise of increasing yields in productive land as well as allowing for cultivation in previously intolerant environments (Fedoroff et al., 2010; Godfray et al., 2010a), potentially resulting in a net biodiversity increase (Carpenter et al., 2011). The protection of existing genetic diversity in the form of wild crop and livestock varieties is key to safeguarding against future environmental change (Mace et al., 2012). Indeed, the presence of wild varietals is essential for isolating yield-boosting genes as well as other desired qualities such as drought and flood resistance (Normile, 2008).

**Box 3.5:** Bioenergy and indirect land-use change

The Global Biosphere Management Model (GLOBIOM) developed by IIASA is used to illuminate the complex interplay of agricultural, bioenergy and forestry production sectors on land-use change. GLOBIOM is a partial equilibrium economic model focused on specific economic sectors (18 most important crops, 7 livestock products, full forestry and bioenergy supply chains) and encompassing 30 world regions in varying degrees of resolution and disaggregation. The model is supported by a comprehensive geospatial database (Skalský et al., 2008) that informs production potential and simulates under a dynamic recursive framework land-use changes at 10 year intervals up to 2100. Indirect GLOBIOM drivers are an exogenous GDP and population growth projections which, together with food consumption per capita (FAO-based), allow for the simulation of supply and demand, commodity markets and international trade. GLOBIOM also represents technological progress in crop and livestock production and land conversion constraints related to biophysical or policy restrictions. Direct drivers are model outputs including spatially-explicit land-use change, GHG emissions, water use, biomass extraction and nutrient balances.

The confluence of bioenergy technologies and government subsidies illustrates the potential for emerging technologies to create new markets with complex synergies and feedbacks. Coupled with market feedback mechanisms, GLOBIOM is capable of modelling a wide range of environmental scenarios and has recently been employed to cast light on the debate surrounding the impact of expanded biofuel production on indirect land-use change (Havlík et al., 2011). The model shows that first generation biofuels (e.g. ethanol and biodiesel) lead to greater deforestation than ‘no biofuels’ under all scenarios and have a negative net effect on global GHG levels through increased indirect land-use change emissions. The adoption of second generation biofuels (derived from woody biomass), produced through existing production forests, leads to the lowest cumulative deforestation as well as the greatest decrease (27%) in overall GHG emissions. Second generation biofuels are thus the most advantageous from the perspective of limiting GHG; however, externalities are highly contingent on the feedstock source, with tree plantations established on cropland and grassland leading to the greatest amount of deforestation and water consumption.

## 3.4 Scenarios and models of direct drivers

Anthropogenic direct drivers are to a significant extent driven by the indirect drivers outlined in Section 3.3. Direct drivers impact biodiversity and ecosystem change at a more proximate level, frequently involving synergies with other direct drivers, and ultimately feeding back into indirect drivers. Salafsky et al., (2008) provides an exhaustive and detailed list of direct threats to biodiversity that broadly fall under the rubric of land-use change, climate change and pollution, natural resource use and exploitation, and invasive species. A general overview of each driver is provided in the following sub-sections, followed by a description of prominent scenarios, models and case studies. As with indirect drivers, direct drivers are subject to differing types of uncertainty and are not equally represented in the existing scenario and modelling literature (Table 3.3).

**Table 3.3:** Degree of uncertainty and utilisation in scenarios and models by direct driver.

| Drivers | Utilisation in scenarios and models | Stochastic uncertainty | Scientific uncertainty | Linguistic uncertainty |
|---|---|---|---|---|
| Land use change | Medium | Medium | Medium | Medium |
| Climate change | High | Low | Low | High |
| Pollution | Low | Low | Medium | Low |
| Natural resource use and exploitation | Low | High | Medium | Low |
| Invasive species | High | High | Low | Low |

## 3.4.1 Land-use change

Habitat modification is seen as a prime driver of biodiversity loss and changes in the level and composition of ecosystem services provided at any given location. Habitat modification is mostly a result of land-use change, either induced by human action or as a result of changes in the physical determinants of the habitat (e.g. due to changes in hydrology or climate). Habitat modification also occurs in marine environments, where trawling has particularly devastating implications for seafloor ecosystems (Hiddink et al., 2006). In most cases, the modification of habitat due to human interference is much faster and more pronounced than changes due to climate change (Lehsten et al., 2015). However, in specific environments such as the arctic tundra region, climate change can also have major impacts on habitat.

Land-use change is the major human influence on habitats and can include the conversion of land cover (e.g. deforestation or mining), changes in the management of the ecosystem or agro-ecosystem (e.g. through the intensification of agricultural management or forest harvesting; see Box 3.6) or changes in the spatial configuration of the landscape (e.g. fragmentation of habitats) (van Vliet et al., 2012; Verburg et al., 2013b).

At the regional scale, a variety of different models have emerged in the past decades to simulate changes in land use driven by demographic change, policies and changing demands for land-based commodities or urban use. Model structure and characteristics are often specific to the scale of application, the research questions and the dominant processes involved. Agent-based models have become popular tools for small areas and when it is important to explicitly represent diversity in land-use decision making (Matthews et al., 2007; Brown et al., 2014). In such models, the changing landscape pattern emerges from the decisions of individual landowners and managers that respond to (often exogenously defined) indirect drivers.

At larger spatial and temporal scales, a simpler conceptualisation of decision making is often applied and land-use change is simulated based on the suitability of locations for a specific land use, with the regional-level demands for the different land uses and spatial constraints resulting from regulations and land-use planning (van Delden et al., 2011). In such models, pixels are the units of simulation and often the state of neighbouring pixels is taken to represent neighbourhood effects and processes such as centripetal forces and economies of scale in urban development. Many global scale land-use models use macro-economic representations of commodity markets and trade simulation in general or partial equilibrium models to simulate land-use change between different world regions. In many cases, land-use decisions are represented by simulating the land-use choice of a representative farm at the regional level (van Meijl et al., 2006) or at the level of coarse spatial units (Schmitz et al., 2012). Spatial patterns of land-use change are calculated using either simple land-allocation algorithms based on land suitability or more complex routines that account for competition between alternative land uses (van Asselen and Verburg, 2013).

Independent of the scale, most land-use models simulate mainly the major conversions of land cover (urbanisation, deforestation, etc.) and ignore the subtler modifications of habitat conditions due to changes in land management and in the spatial configuration of landscapes (Kuemmerle et al., 2013).

This is due to either a lack of fine-resolution data on landscape elements and linear features, or the simplified representation of landscapes by either dominant or fractional land cover (Verburg et al., 2013a).

**Box 3.6:** Agroforestry

High rates of deforestation near biodiversity hotspots are associated with low rates of human development and high population growth, with human development and economic policies emerging as key factors (Jha and Bawa, 2006). Although there is no substitute for primary forest in terms of biodiversity value (Gibson et al., 2011), traditional agroforestry systems foster greater biodiversity than monocrop systems (McNeely and Schroth, 2006) and may serve as one method of ensuring socio-economic livelihoods at the margins of rainforests (Steffan-Dewenter et al., 2007). Agroforestry systems have also been found to reduce dependency on nearby reserves and pristine forests, although economic incentives are important to offset the cost to farmers of planting and maintaining trees on farmland (Bhagwat et al., 2008). Further governance options include the implementation of existing conservation frameworks such as REDD (Reducing Emissions from Deforestation and Forest Degradation) to maximise the conservation of high biodiversity areas (Harvey et al., 2010).

## 3.4.2 Climate change and pollution

**Climate change**

Direct driver pathways of climate change are related to changes in climate and weather patterns impacting in situ ecosystem functioning and causing the migration of species and entire ecosystems. There are indications that climate change-induced temperature increases may threaten as many as one in six species at the global level (Urban, 2015).

Rising atmospheric $CO_2$ concentrations leading to higher ocean temperatures and ocean acidification are expected to have profound effects upon marine ecosystems, particularly coral reefs (Hoegh-Guldberg et al., 2007) and marine communities near the seafloor (Hale et al., 2011). Recent studies projecting reef contraction due to global warming are unanimous in their depiction of the negative impacts on the marine biodiversity that depend on these ecosystems (e.g. Pandolfi et al., 2011), although the direct effects of ocean acidification are highly variable across different taxa (Hendriks et al., 2010).

The construction of climate driver scenarios starts with a forcing on the climate system expressed in irradiance (watts per square meter). For the IPCC Fifth Assessment Report, emissions scenarios consistent with climate forcing targets were constructed as RCPs by a community effort of integrated assessment modelling groups with the aim to inform global circulation models and Earth system models. The biodiversity and ecosystem services-relevant variables characterising RCP scenarios include characteristics of land-use scenarios, which were downscaled to provide spatially-explicit land-use maps for the climate modelling community. Gridded land-use transition data for the past and future time period were developed from the reconstruction based on HYDE 3 agricultural data and FAO wood harvest data and future land-use scenarios from integrated assessment models. These gridded land-use datasets are used as a forcing for some Earth system models participating in the Coupled Model Intercomparison Project experiments, to assess the biogeochemical and biogeophysical effects of land-use and land-cover change in the climate change simulation.

The Inter-Sectoral Impact Model Intercomparison Project (ISI-MIP) used climate change projections to make impact assessments in different Earth system sectors and at different scales. Based on common background scenarios, uncertainties across multiple impact models have been derived. ISI-MIP aims to establish a longer-term coordinated impact assessment effort driven by the entire impact community covering all biodiversity and ecosystem services sectors on global scales and for selected regional and ecosystem-specific case studies. In this way, feedbacks between managing biodiversity and ecosystem services sectors, climate and Earth systems can be studied in a loosely coupled manner. A few groups are currently working on fully coupling all three model types (global circulation models, Earth system models and integrated assessment models), where the latter cover both the climate mitigation and adaptation functions of ecosystem management. Using such full coupling, climate drivers and their biodiversity and ecosystem services feedbacks can be consistently analysed. Decision-support tools can be expected to become more useful in the decades to come, as the temporal (including climate extremes) and spatial resolution of climate signals improve and more transient model runs become available (Fuss et al., 2015).

**Box 3.7:** IPCC scenarios

Global-scale long-run environmental assessments are typically framed in consistency with existing scenario storylines such as the IPCC Special Report on Emission Scenarios (Nakićenoić and Swart, 2000). The scenarios of the IPCC, the MA, the Global Biodiversity Outlook, the Global Environment Outlook and the Global Deserts Outlook have used these storylines or close derivatives of these to generate indirect driver scenarios for their sector-specific outlooks. Regional assessments of the MA and the national variants of the Global Environment Outlook, such as those carried out in the United Kingdom, China and Brazil, have used globally consistent regional variants of existing storylines. Downscaled gridded scenarios of socio-economic drivers of SRES (Grübler et al., 2007) have been used as indirect drivers of forest-cover change (Kindermann et al., 2008). Climate change scenarios are typically provided on the same grid resolution and are used as direct drivers of ecosystem change (e.g. Seidl et al., 2014). Local and more regional specific scenarios of indirect and direct drivers are typically constructed bottom-up and may significantly deviate from the globally established storylines. More recently, associations or even directing mapping of such bottom-up scenarios into global storylines have been performed, allowing for increased comparability across regional case studies (e.g. Vervoort, 2013).

The SRES (Nakićenoić and Swart, 2000), long employed by the IPCC, has given way to a new framework formed by the confluence of the RCPs and the SSPs. RCPs are constructed from radiative forcing targets and present a range of potential futures consisting of a low mitigation scenario, two stabilisation scenarios and one high baseline scenario (Van Vuuren et al., 2011). SSPs, as newly formulated by O'Neill et al., , (2014), illustrate socio-economic factors that would make meeting mitigation and adaptation more or less difficult. Building on previous work integrating SRES with socio-economic scenarios (Abildtrup et al., 2006), this new model takes the form of a dual axis matrix with RCPs representing the possible trajectories of climate change drivers (Moss et al., 2010; Van Vuuren et al., 2011), and SSPs representing possible socio-economic developments that would impact the ability to mitigate and adapt to climate change (Van Vuuren et al., 2012b).

**Pollution**

Pollution is an important driver of biodiversity and ecosystem change throughout all biomes, with particularly devastating direct effects on freshwater and marine habitats. Due to its multifaceted nature, scenario analyses are frequently tailored to the specific subclass of pollution under consideration.

The early reports of the effect of the organochlorine insecticides DDT, along with its analogue DDD, on the western grebe (Garrett, 1977) are one of the most documented examples of the biodiversity-pollution nexus. The end of DDT use in the early 1970s in many countries has already contributed to the recovery of many of the impacted populations. Incidents of the massive killing of marine mammals caused by contamination with polychlorinated biphenyl (PCBs) and other persistent organic pollutants (POPs) that belong to the same organochlorine family were also frequently reported (Kannan et al., 2000; Shaw et al., 2005). More recently, veterinary diclofenac used to treat livestock throughout South Asia has been implicated in the collapse of vulture populations (Oaks et al., 2004), with significant ecosystem services implications (Ogada et al., 2012).

The biodiversity of soil fauna is vital to many ecosystem services, including carbon storage, soil fertility and plant diversity, and insect population control (Wolters, 2001). The degradation of soil biodiversity through industrial pollution can result in the proliferation of invasive and destructive species as well as the loss of endemic microorganisms (Hafez and Elbestawy, 2009). In addition to above-ground plant biodiversity decline, ongoing soil biodiversity loss due to agricultural intensification is likely to impair ecosystem multifunctionality, resulting in decreased carbon sequestration as well as greater nitrogen emissions and phosphorous leaching, among other impacts (Wagg et al., 2014).

At a global level, the atmospheric deposition of nitrogen has been recognised as one of the most important threats to the integrity of global biodiversity (Sala et al., 2000; Butchart et al., 2010). Once nitrogen is deposited on terrestrial ecosystems, a cascade of effects can occur that often leads to overall declines in biodiversity (Bobbink et al., 2010). Within terrestrial biomes, nitrogen deposition through fossil fuels and fertiliser use has been found to impede decomposition and slow microbial growth, with a number of implications for terrestrial biodiversity (Smith et al., 1999; Carreiro et al., 2000; Janssens et al., 2010). Changes in biotic or ecological characteristics are simulated in response to environmental drivers using mathematical representations of the most important processes. Such process-based models are useful for assessing temporal trends and response times. However, they often require a large amount of data for model calibration (Dise et al., 2011).

While terrestrial ecosystems have been affected by nitrogen-phosphorous fertilisers, these have had a far more pernicious effect on the biodiversity of freshwater and marine habitats, leading to eutrophication and hypoxic or ‘dead’ zones that support no aquatic life. Eutrophication and acidification occur when nitrogen and phosphorous – the primary limiting factors for algal growth – are introduced, allowing algal blooms to proliferate which deplete the water of oxygen as well as frequently resulting in toxic algae (Camargo and Alonso, 2006). At a regional scale, various scenario analyses have examined the impact of reduced nutrient loads on coastal ecosystems in the North Sea (e.g. Skogen et al., 2004; Lenhart et al., 2010). Integrated approaches to modelling nutrient emissions have also been conducted on a global scale using the MA storylines and the Global Nutrient Export from Watersheds (NEWS) model, highlighting the role of indirect drivers on future nutrient emissions (Seitzinger et al., 2010).

Plastic debris is emerging as one of the most potent pollutants of marine environments. Results from the ocean circulation model HYCOM (Hybrid Coordinate Ocean Model), coupled with the particle-tracking model Pol3DD, estimate that 5.25 trillion plastic particles weighing 268,940 tons are in the world's oceans (Eriksen et al., 2014). The potential for plastic debris to travel considerable distances, its resistance to biodegradation, and its potential to accumulate in habitats far from its point of origin present a distinct challenge (sCBD, 2012). In addition to the direct introduction of microplastics used in commercial cleaning processes as well as plastic pellets and powders (Barnes et al., 2009), larger pieces of plastic are degraded by the effects of heat, wave action and UV, eventually forming microplastics and nanoplastics ranging from 5μm to 200 nm in diameter (Ryan et al., 2009; Andrady, 2011; Sundt et al., 2014). The ingestion of such plastics by aquatic life can lead to physical blockages, resulting in mortality as well as the accumulation of POPs throughout the food chain (Box 3.8). This problem is particularly pronounced near the ocean floor, where higher density plastics accumulate and are consumed by benthic scavengers which serve as a vector to higher trophic organisms (Wright et al., 2013). In addition to the ingestion of plastic, entanglement in plastic loops and 'ghost nets' affects a number of marine animals, resulting in strangulation and reduced fitness (Derraik, 2002). According to sCBD, (2012), impacts of marine debris have been reported for 663 species.

Plastic pieces also serve as long-lasting vectors of transport across marine environments, introducing invasive species to the detriment of endemic biota (Gregory, 2009). There is also growing evidence that microplastics absorb POPs, serving as a high concentrate vector of transport and ingestion by marine organisms (Teuten et al., 2009). Compounding this phenomenon, climate change has greatly expanded the habitable range of many generalists that are now able to take advantage of such vectors, illustrating the complex interlinkages among biodiversity and ecosystem services direct drivers.

**Box 3.8:** Persistent organic pollutants

POPs are a group of chemicals that include some pesticides, some industrial chemicals, dioxins and furans. The use of POPs has been banned under the Stockholm Convention on Persistent Organic Pollutants, which came into force in 2004 (Ahmed, 2006). The tendency of POPs to dissolve and bioaccumulate in fat tissues, subsequently bioamplifying through food chains, has enabled them to build up in tissues, reaching very high concentrations in organisms at the top of the food chain, causing serious impacts and possible massive death. Recently, various reports have emerged to document the deleterious effect of endocrine disturbing chemicals (EDCs) – a group of chemicals that includes pesticides, industrial chemicals, metals and personal care products – on endocrine systems (Bergman et al., 2013). Other potential pollutants that impact biodiversity include heavy metals (Mulder and Breure, 2006), nutrients (Ochoa-Hueso et al., 2011) and systemic pesticides (Van der Sluijs et al., 2015).

Models have been used to depict changes in ecosystems however, due to the complexity of the biological system, there is little consensus on the basic equations for describing physical systems (James, 2002). As one example, Aquatox is one of the most widely used aquatic ecosystem models. It models chemical fate and effects as a prelude to the evaluation of past and present, direct and indirect impacts of stressors of aquatic ecosystems. Aquatox can simulate flasks and tanks, ponds and pond enclosures, successive stream reaches, lakes, reservoirs and estuaries (Park et al., 2008). The model is frequently used in mapping the bioaccumulation of pollutants in plants, fish and shorebirds that feed on aquatic organisms. However, like most water quality models, Aquatox predicts only the concentrations of pollutants in water but cannot project the effects of said pollutants.

### 3.4.3 Natural resource use and exploitation

The anthropogenic exploitation of wildlife has occurred throughout human history, leading to biodiversity loss and extinctions; however, the recent rate of loss has accelerated sharply (Leakey and Lewin, 1996).

The most overexploited species include marine fish, invertebrates, trees, tropical vertebrates hunted for bushmeat and species harvested for the medicinal and pet trade (MA, 2005b).

As direct drivers of biodiversity and ecosystem change, natural resource use and climate change exhibit interlinkages in the form of climate change-induced increases in scientific and stochastic uncertainty related to the modelling and management of natural resources (Nichols et al., 2011).

Trade in bushmeat is one of the greatest threats to wildlife in the tropics, particularly among large-bodied slow-reproducing species. Indeed, vulnerable species have already been extirpated in many regions, resulting in an 'extinction filter' where the remaining species are those capable of coping with anthropogenic pressures (Cowlishaw et al., 2005). In addition to being a conservation issue, bushmeat hunting and consumption is intricately tied to the livelihood of households not only as a protein source during periods of low agricultural production, but also as a source of income from sales to more affluent urban households (de Merode et al., 2004; Bennett et al., 2007).

There is a general consensus among conservationists that sustainable bushmeat management and harvesting through better regulation is the best available solution to overexploitation, given the socio-economic contexts in many of the affected regions.

Human activities have severely affected ocean health through overfishing, although there are significant country-level differences (Halpern et al., 2012). As the primary driver of the decline in marine resources, the overexploitation of marine habitats has led to precipitous drops in commercially valuable species, as well as other species subject to bycatch and overfishing (Pauly et al., 2002). The decision to exit a declining fishery is highly contingent on the socio-economic status of the fisher, with poorer households less likely to leave (Cinner et al., 2009). Furthermore, there is evidence at the local level that proximity to markets and market demand better predict overfishing than population density (Cinner and McClanahan, 2006). Here, participatory modelling approaches with greater stakeholder involvement at the local level are highly appropriate for applications involving the sustainable governance of natural resources (Videira et al., 2010), with particular salience for the management of fisheries (Röckmann et al., 2012).

Trade in ornamental species, including vertebrates associated with traditional Chinese medicine, has led to significant biodiversity losses, particularly in the South East Asia region (Sodhi et al., 2004; Nijman, 2010). In addition, trade in aquatic ornamental fish serves as a vector for the spread of invasive species (Padilla and Williams, 2004). As a direct driver, natural resource use and exploitation is heavily influenced by indirect drivers such as socio-economic and demographic trends, as well as societal and cultural influences. Indeed, per capita consumption levels are emerging as a potentially more important driver of biodiversity and ecosystem change than population growth (Toth and Szigeti, 2016). Models and scenarios of natural resource consumption and exploitation therefore need to be intimately tied to economic and sociocultural trends.

### 3.4.4 Invasive species

Invasive species may be indigenous and/or exotic/alien, and occur mostly in terrestrial and aquatic ecosystems (marine and freshwater), disrupting the ecological functioning of natural systems. Invasive species outcompete local and indigenous species for natural resources, with negative implications for biodiversity. A number of invasive and alien species or weeds have been reported in various parts of the world, resulting in loss of biodiversity at local and regional scales and causing significant economic damage (Mack et al., 2000).

The type and extent of invasive species will depend on the drivers which, for terrestrial environments, mainly include the type of habitat, soil, climatic conditions and degree of disturbance. The dispersion of invasive species has been extensively studied as a function of both climate and land-use change, with the general finding that climate change is conducive to increased invasions in both terrestrial and marine ecosystems (Hellmann et al., 2008; Rahel and Olden, 2008; Walther et al., 2009). The influence of land-use change is less clear, although habitat type is a good indicator of invasiveness, and disturbed habitats (e.g. arable land, anthropogenic herb stands) are more susceptible to invasion (Chytrý et al., 2008). Most invasive species do not have natural enemies in their new environments and have to be removed using chemical, manual, mechanical or integrated methods.

A number of invasive species-related models have been developed and used in depicting invasive species spread, distribution in new areas, and also for quantifying their impacts on the environment. Climex, first published in the 1980s, is one of the earliest used models of invasive species. The primary output is a mapped prediction of the favourability of a set of locations for a given species, although the model also produces a suite of additional information to allow for a further understanding of species responses to climate. Bioclimatic envelope models such as Climex have been frequently employed to map species distribution, although the predictive accuracy of such models can vary substantially depending on the inclusion of topographic heterogeneity and $CO_2$ concentrations (Willis and Bhagwat, 2009). Spatially-explicit models (Modular Dispersal in GIS, MDiG) were designed as an open source modular framework for dispersal simulation integrated within a GIS (Geographic Information System). The model modules were designed to model an approximation of local diffusion, long distance dispersal, growth and chance population mortality based on the underlying suitability of a region for the establishment of a viable population (Pitt, 2008).

**Box 3.9:** Invasive species in the South African context

Of the approximately 8,750 alien species introduced into South Africa, 161 are seriously invasive, while others have the potential to become invasive in the future (Van Wilgen et al., 2001)**.** In the arid- and semi-arid savannah and grassland biomes of Southern Africa, invasive species occur in areas that are degraded, mostly in rangelands that have been disturbed by overgrazing or mismanagement, negatively impacting the grazing capacity of the area. This thickening of indigenous woody species (also called bush encroachment) is caused by species such as *Senegalia mellifera* (black thorn), *Terminalia sericea* (terpentine bush), *Vachellia tortilis* (umbrella thorn), and *Dichrostachys cinerea* (sickle bush). High-density woody alien species, such as members of the *Prosopis* species (mesquite), compete for moisture with local species, especially in the lower-lying riverine areas and valleys. *Prosopis* invasions in the Northern Cape Province of South Africa result in an estimated water loss of 8.94 million $m^3$ every year.

## 3.5 Lessons learned and the way forward

There are a myriad of models used to make projections of indirect and direct drivers. This diversity reflects the necessity that 'every problem requires its own model' and that one model or model approach alone is unlikely to sufficiently characterise possible futures of drivers and driver processes.

Scenarios and models of drivers often need to be specifically tailored to the needs of different policy or decision contexts. Existing approaches can be useful for the data they contain, but rarely deliver meaningful results or even insights if applied without proper adaptation to a particular decision context. There is no single scenario development or modelling tool that serves the needs of the full range of application domains. Even integrated assessment and general equilibrium models, in and of themselves, typically fall short of capturing the necessary details required by biodiversity and ecosystem services applications.

However, although integrated assessment models or general equilibrium models will rarely be the recommended model of choice for a specific biodiversity and ecosystem services study, they may still be indispensable for providing boundary conditions. Linking the macro-model context to specific biodiversity and ecosystem services models will ensure globally consistent local results and sector-specific consistency in a wider socio-economic context.

Given that the science of developing driver scenarios is still maturing, the way forward will require an increased focus on refining strategies to improve the characterisation of uncertainties, including notions of ignorance, through improved creativity in building scenario storylines to better characterise the possibility spaces of driver sets and their evolution over time. Uncertainty can be elucidated by identifying and eliminating bias, and by increasing precision through making models more data-driven where robust data are available. Model bias is mainly related to spatial, sectoral and temporal inconsistencies. Strategies for addressing these (and discussed further in Chapter 6) include:

- Clusters of spatially linked models need to be developed to guarantee the relevance and consistency of scenarios of biodiversity and ecosystem services change from the global to the local level. The two-way spatial coupling of models in combination with hierarchically nested scenario storyline building will ensure that local case studies are consistent with global assumptions and, at the same time, that the upscaling of local knowledge can enrich storylines on larger spatial aggregates (Verburg et al., 2015).
- Interactions of biodiversity and ecosystem services with the wider socio-economic system will need to be modelled through appropriate response functions or through direct or indirect model linkage with high-resolution driver information needed for a specific biodiversity and ecosystem services study and more aggregated models covering the rest of the socio-economic system.
- In many cases, environmentally-defined spatial scales and units of analysis would be more relevant for biodiversity driver scenarios than other scales and units (e.g. administrative, municipal, provincial or country). Laura, (2009) assesses the challenges of conserving biodiversity across the US-Mexican border, finding that many problems are often exacerbated by socio-economic and cultural differences. This study shows how access to relevant information on biodiversity drivers is particularly affected when ecosystems are artificially divided by different administrative regimes. In these cases, information-sharing tends to be slow, policymaking processes can be delayed, and key options for protecting shared resources tend to be overlooked.
- The issue of temporal inconsistencies has a long-standing history in natural resource management, since the introduction of discounting in forest management by Faustmann, (1849), and is a strong driver of human-impacted ecosystem change and driver management. Harmonising long-term

strategies with short-term actions remains a challenge. Forecasting tools for short-term market variables will need to be connected to projection tools carrying out long-run analyses of market and environmental resource variables.

Improvements in the precision of existing tools will necessitate the assimilation of large amounts of Earth observation data, market information and observations describing dimensions of human behaviour and human capital, including knowledge of biodiversity and ecosystem services management (see Chapter 8). Data-driven approaches to precision improvements will need to be applied to identify parameters of scenario models.

Scenario storyline formulation for indirect and direct drivers has a long tradition in foresight studies, economic analysis and demographics, and more generally in integrated assessment and impact assessment. Most scenario assessments are of a deterministic nature and typically ask the question what the best policy options would be given a single driver reference scenario. While some biodiversity and ecosystem services studies can be ‘pegged’ to existing driver scenarios or scenario families, in many circumstances new scenarios of indirect and direct drivers departing from existing global environmental assessment scenarios such as IPCC SSPs/RCPs and MA will need to be constructed to find a better fit with biodiversity and ecosystem services-specific contexts. In this case, existing scenarios will serve as reference points and benchmarks for specific biodiversity and ecosystem services driver scenarios. Due to the long-lasting nature and irreversibility of many biodiversity and ecosystem services-related decisions, the current practice of operating with only one reference driver scenario needs to be augmented by developing multiple reference scenarios entering decision making under uncertainty tools, which will ensure that biodiversity and ecosystem services management strategies are robust under a wide range of possible driver scenarios, or at least allow for the transparent assessment of relative risks.

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# 4 Modelling impacts of drivers on biodiversity and ecosystems

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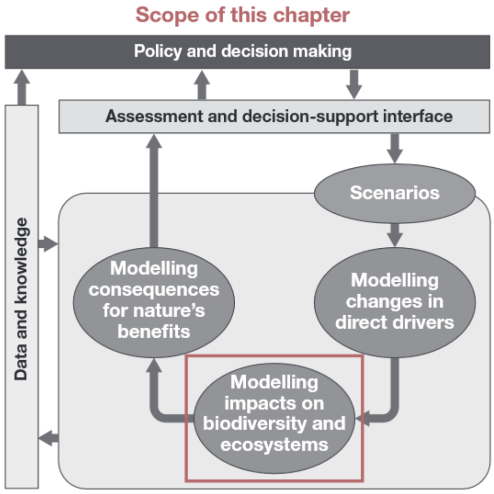


**Purpose of this chapter:** Explores key issues in modelling impacts of changes in direct drivers (from Chapter 3) on biodiversity and ecosystems; and critically reviews major types of models for generating outputs that are either directly relevant to assessment and decision-support activities, or are required as inputs to subsequent modelling of nature's benefits to people (covered in Chapter 5).

**Target audience:** Aimed mostly at a more technical audience, such as scientists and practitioners wanting to identify appropriate biodiversity and ecosystem modelling approaches for particular applications.

## Key findings

**Models of biodiversity and ecosystem function are critical to our capability to predict and understand responses to environmental change (Section 4.2).** Modelling is one way of helping policymakers assess the impacts of different drivers on biodiversity and ecosystems, as well as the feedbacks on drivers generated by those impacts (from Chapter 3). Modelling provides tools that can help project future dynamics based on scenarios of direct and indirect drivers. However, the capacity of biodiversity modelling to meet policymaking needs is still affected by a lack of data and knowledge, and by model complexity and uncertainties.

**There is a need to match biodiversity and ecosystem function model development to stakeholder and policy needs (Section 4.3.2.1).** Biodiversity and ecosystem models rely heavily on assumptions about key processes and input data. There is a need to involve both stakeholders and modellers in representing these processes and assumptions and in identifying critical drivers (i.e. outputs from scenarios, Chapter 3) and the biodiversity/ecosystem response variables that need to be addressed. It is also important that the underlying context, uncertainties, validity, specificity, and outputs of the models are clearly and transparently interpreted and explored jointly by the modellers and stakeholders.

**Biodiversity and ecosystem modelling depends heavily on our understanding of ecosystem structure, function and process and on their adequate representation in models (Section 4.2.1).** Both understanding and adequate representation depend on data availability, so there is a need to generate and compile representative data for different biodiversity variables in different ecosystems at multiple locations and different scales. Observation networks, as well as long-term monitoring programmes, are therefore critical to assess the response of ecosystems to drivers of change such as climate change, land-use change, exploitation and pollution, and to inform model development.

**Uncertainty in ecosystem dynamics is inherent in ecosystem modelling (Section 4.6).** Uncertainty, which is inherently associated with model processes, data limitations and environmental stochasticity, can be accounted for by using multi-model ensembles, quality assurance and quality control, and by generating data from long-term observations. Different models may provide results that should be interpreted within the context of the model assumptions and input data. The biodiversity and ecosystem functioning models currently available provide a range of options to assist policymakers in understanding relationships between drivers and impacts, and in designing efficient policies.

The scientific community has recognised the importance of developing strategies to address the limitations of current models and of suitably treating the different sources of uncertainty involved. Well-established guidelines are relevant because an assessment of available approaches to modelling biodiversity and ecosystem responses to environmental changes clearly concludes that there is no single modelling approach (or model category) that can serve all assessment needs and decision-making requirements.

## Key recommendations

**Efforts should be made to ensure that experts involved in Intergovernmental Platform on Biodiversity and Ecosystem Services (IPBES) deliverables are aware of the important role that models of biodiversity and ecosystem functioning play in formalising the complexity of living systems (Section 4.2).** In particular, it is important that experts in IPBES task forces and assessments recognise the complexities linking drivers of environmental change to ecosystem dynamics. It is also important that they acknowledge the value of modelling as a method to formally represent – and therefore simplify – such complexity, and as a scientific tool to support decision making. This can be facilitated by the selection of experts for IPBES deliverables who are familiar with the limitations and use of models of biodiversity and ecosystem functioning. In addition, follow-up activities in Deliverable 3c can provide additional guidance to experts in IPBES deliverables – especially the thematic, regional and global assessments – to assist in the interpretation and correct use of biodiversity models.

**Encouraging stakeholder participation in scenario and model use as early as possible in assessments would provide substantial benefits for IPBES (Section 4.7).** This would maximise the correspondence between the objectives of the assessments and the outputs and limitations of the ecosystem modelling approach to be developed or interpreted. It is important that modellers and stakeholders interact in the different stages of modelling exercises concerning the selection of key questions, the context, assumptions, scale, time frame, and so on. Mechanisms for facilitating this dialogue are not yet explicitly laid out in the IPBES Work Programme.

**Experts involved in IPBES assessments should critically evaluate the quality of the information used in modelling exercises. The task force on Knowledge, Information and Data could encourage long-term observations that would improve our understanding of biodiversity and ecological patterns (Section 4.3.2).** This will enable models and outputs to be improved and to better fit IPBES objectives. IPBES needs to ensure that a quality chain between data type, model output and suitability for end-use exists in all assessments. Linkage of these components cannot be adequately implemented if data are scarce or of a low quality, thus leading to constraints in how model outputs feed into a given decision context (Chapter 8).

**The development of consistent protocols is important for IPBES to ensure the quality of the use of models and their outputs in assessments (Section 4.3.2.2).** Model intercomparison programmes would encourage increased collaboration among the modelling groups and with field ecologists to develop suitable protocols for modelling drivers impacting on biodiversity and ecosystem functions. An example could be to engage the scientific community to form model intercomparison groups similar to those developed in the context of the Intergovernmental Platform on Climate Change (IPCC) assessments, involving a large number of modelling groups working on biodiversity and ecosystem modelling.

**The explicit characterisation of uncertainty should be a priority in the presentation and use of biodiversity and ecosystem model outputs within IPBES (Section 4.6).** Communication of outputs needs to adequately identify the uncertainties associated with scenario development (Chapters 2 and 3), as well as clearly describe and communicate issues directly related to the choice of biodiversity and ecosystem models. To enable robust decision making and to account for uncertainty in the outcomes of biodiversity models, the integration of multi-model techniques and ensembles of multiple models and scenarios that provide a range of projections could be promoted in assessments. These practices should be encouraged, including by engaging with the scientific community through the task force on Knowledge, Information and Data and through the follow-up activities of Deliverable 3c.

**The development of guidelines for integrated ecosystem modelling would be highly beneficial for IPBES assessments.** There is a need to develop integrated models that can be applied in marine, terrestrial and freshwater ecosystems to assess the impact of drivers and their feedbacks on biodiversity and ecosystems. These integrated models should consider both biophysical and socio-economic drivers and their feedbacks at scales relevant to ecological processes underlying biodiversity changes and to decision-making processes.

# 4.1 Introduction and conceptual framework

Biodiversity and ecosystem dynamics are inherently complex, and so is their response to environmental drivers – including both natural and anthropogenic drivers. Models are powerful tools for addressing complex systems as they can be used to assess and predict the impacts of drivers on biodiversity and ecosystems, and hence the impacts on ecosystem services and human well-being. This chapter focuses on the approaches and methods currently available to explicitly link environmental changes with biodiversity and ecosystem responses, from changes in population size, to community composition and structure, to biogeochemistry fluxes. The aim is to identify the range of tools available for unravelling patterns and mechanisms of biodiversity and ecosystem change, and to incorporate this knowledge in models, allowing the projection of the future state of biodiversity and ecosystems in particular decision-making and management contexts (see Chapter 2).

The chapter first provides an introduction to the context in which biodiversity and ecosystem models are to be developed, including the relevant aspects of biodiversity response to drivers and a typology of the main modelling approaches (Section 4.2). Next, an overview of available modelling approaches relevant to IPBES – at different levels of biological organisation – is provided (Section 4.3). This comprises an explanation of model structure, scope of application and illustrating examples. To further guide the use of the most appropriate models, this section includes a critical analysis of the different modelling tools available, of model limitations, and of existing information and capacity-building needs.

Sections 4.4 to 4.6 cover the main issues in biodiversity modelling, which are modelling biodiversity feedbacks into environmental drivers, balancing model complexity and applicability, and addressing uncertainty. The issues associated with sources of uncertainty in model projections are of the utmost importance in the context of biodiversity projections for IPBES, and we describe this topic in depth in the context of biodiversity and ecosystem modelling. Finally, we identify the major challenges to biodiversity projections in the context of the IPBES programme, and highlight the main pathways available to policymakers at a range of administrative scales.

This chapter is directly linked to Chapter 3 (scenarios and models of indirect and direct drivers) and to Chapter 5 (modelling nature's benefits to people). The models discussed in this chapter provide a means of translating scenarios of drivers, as described in Chapter 3, into expected impacts on biodiversity and ecosystems. In turn, outputs (i.e. projections) from the models described in this chapter can serve as inputs to modelling changes in nature's benefits to people (including ecosystem services), as discussed in Chapter 5. Moreover, because the engagement of stakeholders in biodiversity modelling exercises and the effective communication of results to policymakers are fundamental to the successful use of models, there is a two-way link between the present chapter and Chapter 2.

The main external input when modelling biodiversity response to environmental change or pressures is the change in the state of drivers directly affecting biodiversity and ecosystems. In this chapter, we consider modelling approaches that assess the impacts of direct drivers of environmental change as identified by the Millennium Ecosystem Assessment (MA, 2005a): habitat change, climate change, overexploitation, pollution and invasive species. Scenario development and modelling methods for projecting future changes in direct drivers, to be used as inputs in biodiversity and ecosystems models, are described in detail in Chapter 3.

As for connections and potential overlap with Chapter 5, it is important to note the multiple roles of biodiversity and ecosystems in the conceptual chain linking direct drivers to nature's benefits to people. Specifically, biodiversity may either regulate the ecosystem processes that generate final ecosystem services, or itself constitute a final ecosystem service, or even provide a good that is directly enjoyed by people (Mace et al., 2012; Oliver et al., 2015). In the first case, biodiversity attributes affect the development and maintenance of ecosystem processes (Cardinale et al., 2012), such as nutrient cycling (Handa et al., 2014), primary productivity (Cardinale et al., 2007) or water infiltration (Eldridge and Freudenberger, 2005), which in turn give rise to final ecosystem services. In the second case, biodiversity elements are themselves material outputs with direct use value, such as medicinal plants or fish, but require human capital inputs (e.g. labour, transport) before being enjoyed by society. Finally, biodiversity elements may themselves be viewed as a good if directly enjoyed by people without any additional input, which is the case with the aesthetic enjoyment of nature, ecotourism, and so on. Therefore, outputs from biodiversity models (including future projections) can be used as inputs to ecosystem services models, or provide direct information on ecosystem services and goods, such as data on the distribution and abundance of charismatic species. It is worth noting that, often, ecosystem services models implicitly (e.g. by simplifying biodiversity components and ecosystem functions using surrogate information on land cover or use) or explicitly include biodiversity or ecosystem function sub-modules. A compilation of relevant cases is treated in further detail in Chapter 5. Moreover, although biotic and abiotic ecosystem components interact and are both essential to ecosystem functioning and therefore to modelling ecosystem services – in particular regulating services – the focus of this chapter will be on the biotic components, represented by 'nature' in the IPBES Conceptual Framework (see Figure 1.2 in Chapter 1).

In accordance with the overall aim of Deliverable 3c to inform and guide other IPBES deliverables in the use of scenarios and models for biodiversity and ecosystem services, this chapter provides relevant information on:

- Modelling methodologies available for the IPBES Catalogue of Policy Support Tools for assessing the response of biodiversity and ecosystems to direct drivers (Deliverable 4c);
- Available modelling methodologies to evaluate scenarios of sustainable use of biodiversity and to assess responses to drivers of land degradation and to invasive species (Deliverable 3b);
- Caveats and good practices for assessments regarding the use of available data in modelling approaches and the use of modelling outputs in literature reviews and meta-analyses (Deliverables 2b, 2c, 3b);
- Capacity-building needs regarding the use of modelling approaches in decision-making processes and the engagement of stakeholders in modelling processes (Deliverables 1a, 2b);
- Current knowledge gaps, data needs and future research recommendations to improve the predictability and scope of application of models (Deliverable 1d);
- Involving indigenous and local knowledge in model development, testing and application (Deliverable 1c).

## 4.2 Structure and components of biodiversity and ecosystem models

Scientists and stakeholders supporting decision-making processes are always faced with the challenge of selecting the key processes and drivers leading to relevant impacts on their study object (Guisan et al., 2013), and this is the topic of this section. Decisions on how and what to include explicitly in the modelling process, and what can be simplified or ignored, are crucial as they will impact model outcomes. The role of biodiversity as a regulator of ecosystem processes or as a material output (either a final service or good) defines the variables of interest when assessing and projecting the impacts of direct drivers. For instance, community data such as functional or species diversity (Cardinale et al., 2007; Mace et al., 2012) or habitat structure (Eldridge and Freudenberger, 2005) may be particularly important in assessing the impact of drivers when biodiversity has a regulatory role, while population data such as species distribution (Gaikwad et al., 2011) or population structure (Berkeley et al., 2004) would be more appropriate when biodiversity elements have a direct use value. It is also worth noting that, overall, a positive relationship exists between biodiversity attributes and ecosystem services (Harrison et al., 2014).

This recognition of the different roles of biodiversity follows an anthropocentric perspective that focuses on ecosystem services – the material and non-material benefits generated by nature. Like utilitarian values, biodiversity has its own intrinsic value that is independent of human demand or appreciation and that is difficult, or even impossible, to quantify through modelling, although its existence or evolutionary value may serve to maintain life.

Biodiversity models, like other mathematical models in the environmental sciences, consist of a set of components, namely state variables, external variables, mathematical equations and parameters (Jørgensen and Bendoricchio, 2001; Smith and Smith, 2007). Predictions of ecological responses to environmental changes should start with the specification of the major conceptual components of the model and the critical relationships between them. In the description of any model of this type, the following components should be identified:

1. **Elements describing the ecosystem characteristics.** These are the target state variables used to describe the biophysical components of interest, such as biomass, species richness, functional diversity or habitat structure (see Figure 4.1). State variables should be included based on their ability to serve as indicators of system state, their sensitivity to pressures, and the stability of their response pattern, although the consideration of available versus ideal data often calls for a pragmatic approach given the costs and feasibility of data collection.
2. **Environmental and biotic drivers.** The spatial or temporal dynamics of these model components have a direct or indirect effect on the state variables. In the context of environmental change, changes in the value of environmental (e.g. climate change) and biotic drivers will affect the value of the state variables (e.g. species distributions).
3. **Ecosystem/ecological processes.** These model components allow the description of the changes in the stock and/or flow of materials or in the interactions between organisms and with their abiotic environment (Mace et al., 2012). Processes are relevant in determining changes in the biological component (e.g. changes in species distribution after colonisation and extinction dynamics).

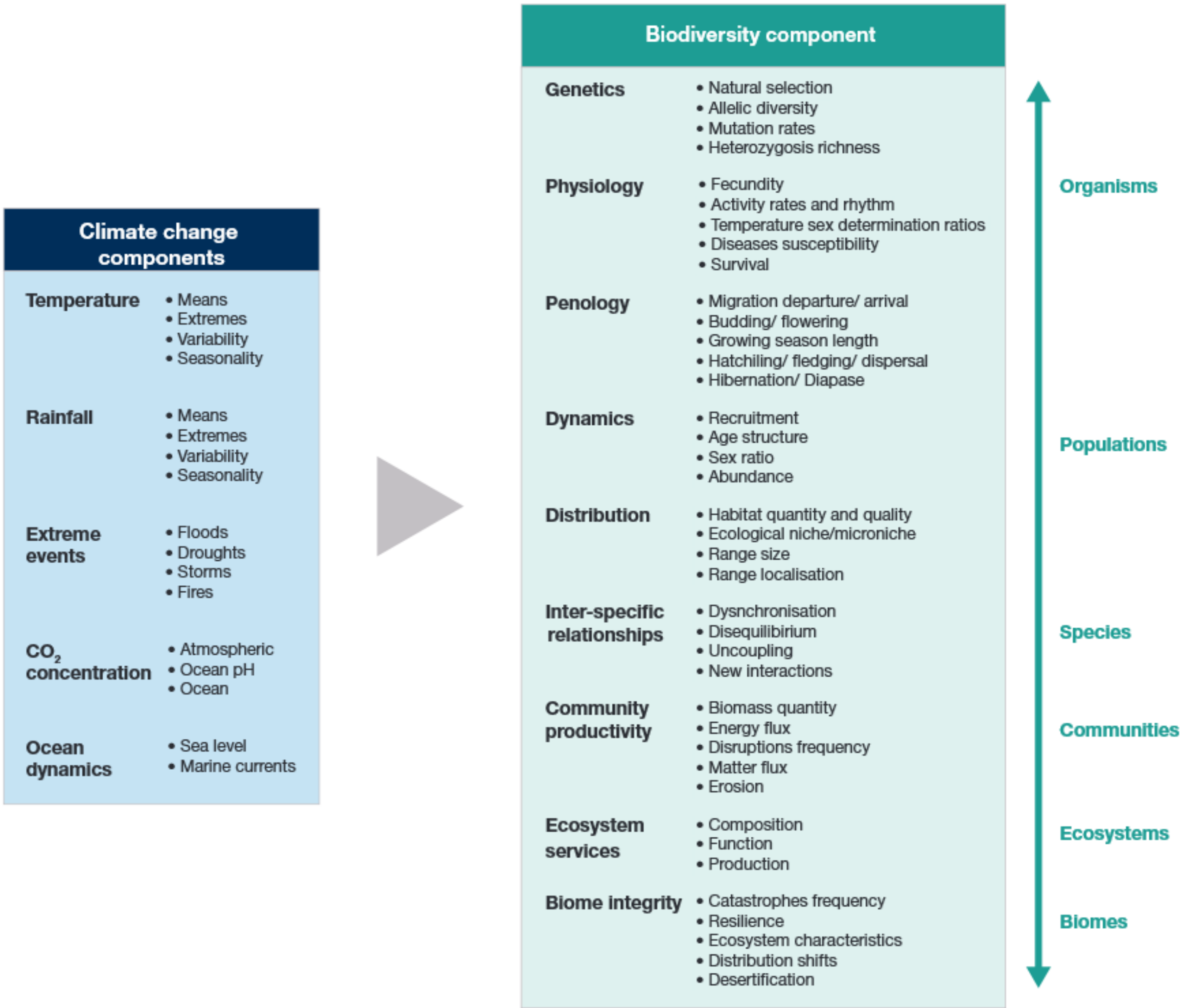


**Figure 4.1:** Summary of biodiversity state variables and processes affected at different organisational levels by different components of climate change (Modified from Bellard et al., 2012. *Impacts of climate change on the future of biodiversity.* Copyright © 2012 by John Wiley Sons, Inc. Reprinted by permission of John Wiley & Sons, Inc).

The impact of drivers on biological processes is key in determining the nature of the model and the inclusion of multidisciplinary expertise in the model-building process (Guisan et al., 2013). In the context of environmental change, the effect of environmental pressures on state variables can be direct (e.g. loss of tree cover after deforestation, changes in climate conditions) or mediated by biophysical processes (e.g. ocean acidification and warming affecting coral recruitment and growth, and hence coral abundance and reef structure). In addition, processes also mediate interactions among state variables (e.g. biotic interactions, trophic cascades).

Using community structure as an example, the processes and scales that are important for modelling are illustrated in Figure 4.2, which shows how ‘filters’ select species from a global pool to obtain realised local communities (Thuiller et al., 2013). In other words, and in the context of biodiversity response to change, drivers (input data) create or change geographic or niche filters, thus leading to changes in community composition (output data). The filters (ecological processes involved) include biogeographic and environmental aspects of the real world, and are represented as components in biodiversity models. Species response to direct drivers (box a) is mediated by dispersal and niche filters through a series of processes (box b), which may or may not be explicitly considered in biodiversity models.

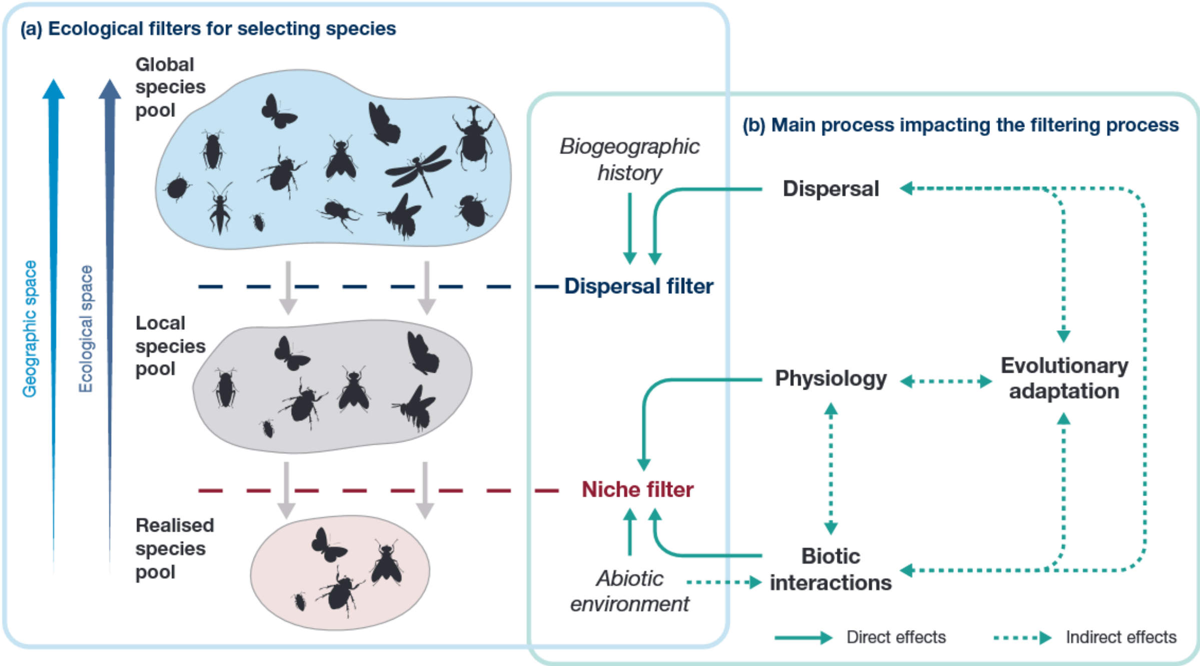


**Figure 4.2**: (a) Conceptual diagram of how dispersal and niche 'filters' select species from pools at different geographical and ecological scales. (b) Main processes that directly or indirectly impact the filtering process (Modified from Thuiller et al., 2013. *A road map for integrating eco-evolutionary processes into biodiversity models.* Copyright © 2013 by John Wiley Sons, Inc. Reprinted by permission of John Wiley & Sons, Inc).

### 4.2.1 Describing ecosystems in models: biological levels for modelling

Biodiversity and ecosystem responses to environmental change can assume many forms as a consequence of the inherent complexity; one way of addressing this diversity is to reduce it to a few meaningful dimensions. Biodiversity and ecosystem variables can be arranged along dimensions representing key aspects of biodiversity complexity: biological organisation levels (species, populations, ecosystems, etc.) and biodiversity attributes (composition, structure and function). These two dimensions define a conceptual space that can be useful for identifying relevant response variables (see Table 4.1). More specifically, composition and structural elements such as species richness or biomass correspond to state variables, and functional elements such as primary productivity, herbivory or competition correspond to processes. Composition and structure emerge from processes, but also affect them (Dale and Beyeler, 2001).

From an ecological perspective, composition and structure variables describe the structural elements of ecosystems, while processes describe the fluxes of energy and matter and the interactions within and between organisation levels.

Ecosystems are open systems. They harness solar energy and transfer it through their various structural elements and organisation levels, via different biological and ecological processes. At the biosphere level, water and nutrients (e.g. carbon, nitrogen and phosphorus) are key structural elements of all living components, and key abiotic components of ecosystems. Their flux across the Earth system is described by the biogeochemical cycles. This flux of energy permits life on Earth and fuels the ecological functions that are useful for societies (i.e. ecosystem services). To model the dynamics of biodiversity, it is important that the major ecological processes involved in the transfer of energy through ecosystems are taken into account (Mokany et al., 2015). These include production, consumption, respiration and recycling. Other processes such as regulation and evolution are critical to the maintenance of biodiversity and the resilience of ecosystems over time.

Primary production and respiration are major ecological processes, occurring at the organism level but affecting population dynamics and community structure. Organic matter from primary production forms the basis of all life on Earth. Numerous factors such as light, the availability of inorganic nutrients, water and temperature influence primary production. Respiration, which encompasses all the living processes using oxygen, is at the core of metabolism. While occurring at the organism level, both processes can be considered at every level of organisation. Primary productivity, for instance, is often used as an indicator of ecosystem functioning and modelled at the level of communities or ecosystems to assess the impacts of land-use change, climate change and management practices on vegetation. Regarding respiration, at the organism level respiration processes are influenced by many factors, including the species considered (body-size scaling rules imply that many metabolic processes vary with the maximum size that a species can reach (Kearney et al., 2010), the size of individuals, their condition, the availability of food, oxygen levels and temperature. At the population level, respiration integrates the metabolism of all individuals. It is therefore highly dependent on the size and state structure of the population. At the community level, respiration integrates the metabolism of all populations and is therefore controlled by their relative abundance and the structure of the community. Consumption and recycling are the main processes associated with trophic interactions, and are therefore modelled at the community and ecosystem levels (Sarmiento and Gruber, 2006). Consumption constitutes a major process of ecosystem dynamics that transfers solar energy along food chains, from primary producers up to top predators. Trophic interactions are influenced by various factors, including the spatial-temporal co-occurrence of grazers/predators and their food/prey, which is often constrained by environmental features.

In addition to the metabolic processes described above, processes related to biodiversity responses to environmental changes can be broadly divided into population and community responses (Lavergne et al., 2010). The first of these are mechanisms related to the ecology of the species populations, including dispersal, plasticity and population dynamics. These processes are primarily determined by biological traits expressing the capability of the target species to deal with environmental variability in space and time (e.g. Thuiller et al., 2013; Hanski et al., 2013). Secondly, species interactions can restrict or expand the set of places that the species is able to inhabit (Davis et al., 1998). Competition, facilitation or trophic relationships are site- and species-specific and account for a great deal of variability in the capability of a species to survive in a given environment.

**Table 4.1:** Examples of biological levels for modelling (compositional, structural and functional biodiversity variables, from (Noss, 1990; Dale and Beyeler, 2001), selected to represent levels of biodiversity that warrant attention in environmental monitoring and assessment programmes.

| Level | Composition | Structure | Function |
|---|---|---|---|
| Individuals | Genes | Genetic structure | Genetic processes, metabolism |
| Populations | Presence, abundance, cover, biomass, density | Population structure, range, morphological variability | Demography, dispersion, phenology |
| Communities | Species richness, evenness and diversity, similarity | Canopy structure, habitat structure | Species interactions (herbivory, predation, competition, parasitism), decomposition |
| Ecosystems | Habitat richness | Spatial heterogeneity, fragmentation, connectivity | Ecosystem processes (hydrologic processes, geomorphic processes), disturbances |

### 4.2.2 Introducing drivers of environmental change

The world has experienced global environmental change due to human activities, and this has encouraged research on scenarios and models to study the new challenges that biodiversity is exposed to (Pereira et al., 2010). Assessments of links between these drivers and biodiversity responses are central to IPBES. Change in biodiversity is determined both by changes in the environment and by the ecological and physiological processes contributing to the dynamics of these ecological systems (Lavergne et al., 2010). Thus, biodiversity change may be either related to changes in the environment itself, to the biological processes acting within ecosystems or, more frequently, to a combination of both (Leung et al., 2012). It is therefore important to distinguish between changes caused by anthropogenic drivers and changes emerging from the natural dynamics of ecological systems. This is particularly important because, although biodiversity and ecosystem services experience change due to natural causes, anthropogenic drivers increasingly dominate current environmental changes.

Following the IPBES Conceptual Framework, natural and anthropogenic drivers directly affect biodiversity. Both natural and anthropogenic direct drivers of impacts on ecosystem processes explicitly cause measurable changes in ecosystem properties.

> Natural direct drivers emerge from natural biophysical and geophysical processes, while anthropogenic drivers result from the trajectory and interactions of socio-economic drivers (indirect drivers).

Biodiversity models use variables describing properties of direct drivers as inputs to predict their impact on biodiversity variables. Historically, the largest impacts on biodiversity have been through land-use change in terrestrial ecosystems (Pereira et al., 2012) and through resource exploitation in marine ecosystems (MA, 2005b). Freshwater ecosystems have been strongly impacted by a range of factors including, most notably, habitat modification, invasive species and pollution. Climate and land-use changes have probably now reached a similar level of pressure on ecosystems, but during the last three centuries land-use change has exposed 1.5 times as many landscapes to significant modifications as climate change (Ostberg et al., 2015).

Human impacts on the global environment are operating at a range of rates and spatial scales. Scaling issues are particularly important when assessing impacts on biodiversity and ecosystem services because drivers have different impacts at different scales. For example, while climate change is a driver that acts at the global scale, habitat modification has an impact on biodiversity and ecosystem services at regional and local scales. The consequences of habitat modification have been significant for many aspects of local, regional and global environments, including the climate, atmospheric composition, species composition and interactions, soil condition, and water and sediment flows. However, global-scale assessments typically mask critical sub-global variations, thus underestimating the effects of drivers acting at local scales. Local and regional case studies can provide the spatial and temporal resolution required to identify and account for major environmental sources of variation in cause-to-cover relationships and the consequence for biodiversity. Single-factor explanations, at the macro or the micro scale, have not proven adequate (Bellard et al., 2015). Many models assessing the impact of environmental drivers on terrestrial ecosystems and biodiversity elements, including those dealing with climate and trace-gas dynamics, require projections of land-cover change as inputs. In this context, Loreau et al. (2003) highlighted that knowledge of spatial processes across ecosystems at the local scale is critical to predict the effects of landscape changes on both biodiversity and ecosystem functioning and services.

### 4.2.3 Dealing with processes: the model continuum from correlative to process-based approaches

There are a wide variety of ecological models available for assessing impacts of direct drivers on biodiversity and ecosystem functioning. These can be categorised based on their complexity and degree of formalisation, from expert-based systems that rely on experience (including in the form of local knowledge), to complex integrated ecosystem models.

Quantitative models are generally classified in two broad categories: correlative and process-based models (e.g. Pereira et al., 2010; Dormann et al., 2012). To distinguish between these model types, we follow the model definitions of Dormann et al. (2012). These state that correlative models are characterised by having parameters with no predefined ecological meaning, and for which processes are implicit, whereas process-based models use explicitly-stated mechanisms, and their parameters have a clear, predefined ecological interpretation.

In the literature, the terms process-based model and mechanistic model are often used as synonyms. Here, we use the term process-based for any model type with explicit implementation of ecological processes in the model implementation (i.e. encompassing both process-based and purely mechanistic models), and we reserve the mechanistic category for the subset of models that are developed based on ecological theory only and that do not use correlative approaches at all for parameterisation. The primary difference along this modelling axis is the inductive versus deductive approach to processing information. The main advantage of correlative models, also termed phenomenological or statistical models, is that there is no need for a fundamental understanding of the ecosystem and relationships between system elements, as these are derived inductively from empirical observations. With process-based models, there is a deductive process involved in which the process is determined and the relationship derived, quantified and explicitly modelled (Jørgensen and Bendoricchio, 2001). At the other end of the formalisation gradient, pure mechanistic models – also called theoretical models – are axiomatic constructions (Gallien et al., 2010). As in theoretical physics, they apply the hypothetico-deductive scientific method, starting from a hypothesis (the axiom) to deduce predictions that can be tested empirically, either to falsify or conversely to corroborate the hypothesis made (but never to prove or 'validate' it).

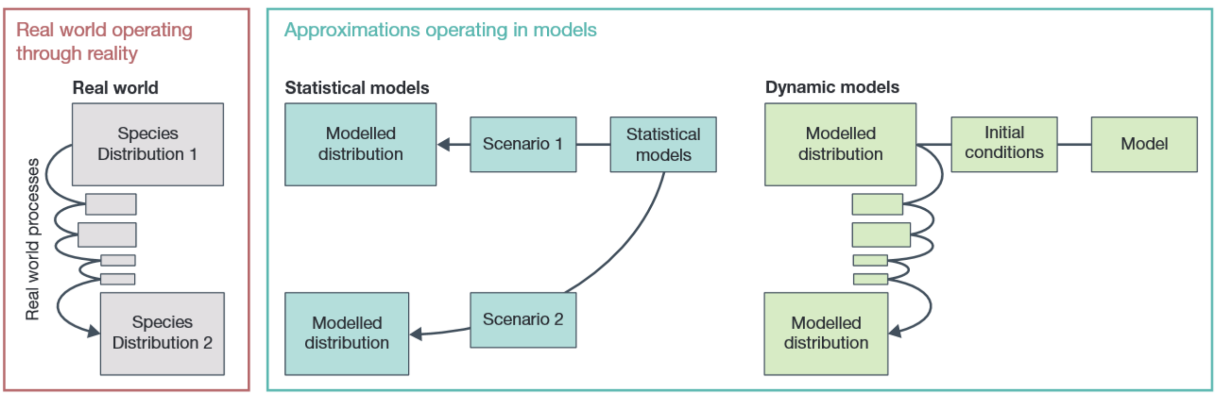


**Figure 4.3**: Schematic representation of the relationship between two observations of a species distribution in the 'real world', 'correlative (statistical) models' and 'dynamic, process-based models' (Modified from McInerny and

To illustrate how models are both abstractions and representations of reality, Figure 4.3 shows how real-world processes change an entity (here a distribution) from one state to another. In a correlative model, the two distribution states are modelled with two alternative scenarios (e.g. before and after a forest fire). In the process-based dynamic model, the model builds on a set of initial conditions to derive a modelled distribution, which then is altered through specified processes that aim to replicate the real-world phenomena in order to predict the second modelled distribution. It must be noted that the real-world processes are often unknown and indeed never can be fully known or emulated. Process-based modelling therefore cannot be expected to fully replicate the real-world situation, but it may provide a useful approximation (McInerny and Etienne, 2012).

In practice, the categorisation of ecological models is rarely as clear-cut as depicted in Figure 4.3, but rather tends to fall along a continuum from correlative to process-based, depending on available data and parameters, purpose and model philosophy. This model continuum, however, forms the basis for the presentation here, which also describes a spectrum of how the broad model types rely on empirical data versus ecological knowledge.

Whether modelling is based on correlative or process-based approaches (or any intermediate type), there are a number of issues that should be considered as part of the model building process (Table 4.2). For instance, statistical assumptions about error structure and unbiased sampling apply to both broad types of modelling approaches. The same is not true regarding the assumption that species are in equilibrium with their environment, which applies only to correlative models, at the risk of losing predictive ability.

#### 4.2.3.1 Expert-based systems

The most common approach for evaluating impacts of alternative management procedures related to predictions and decision support is often based on information provided by experts (Cuddington et al., 2013). An expert is defined here as someone who has achieved a high level of knowledge on a subject through his or her life experience (Kuhnert et al., 2010), and may be a person with local knowledge or a scientist. It is assumed that the expert is a reliable source of information in a specific domain, though it appears that experts tend to be far more confident in their opinions than is warranted (Burgman, 2005).

Eliciting expert information usually involves dealing with multiple expert judgements, with different sources of bias and uncertainty around expert estimates (Martin et al., 2012). For instance, expertise may vary geographically, with relevant information restricted to the region of interest of the experts (Murray et al., 2009). Structuring how multiple expert opinions are used, for example through a Delphi approach (MacMillan and Marshall, 2006), can make the modelling much more rigorous and less likely to result in arbitrary predictions (Sutherland, 2006).

The expert-based approach typically includes five steps: considering how the information will be used; deciding what to elicit; designing the process for the elicitation; the actual undertaking of the elicitation; and finally translating the elicited information into quantitative statements that can either be used directly or in an integrative or participatory modelling approach (Martin et al., 2012).

Expert knowledge-based species-habitat relationships are used extensively to guide conservation planning, particularly when data are scarce (Iglecia et al., 2012). Expert knowledge is quite commonly utilised in conservation science (Janssen et al., 2010; Aizpurua et al., 2015), and has frequently been incorporated in aquatic habitat suitability modelling to link environmental conditions to the quantitative habitat suitability of aquatic species (Mouton et al., 2009).

**Table 4.2**: Summary of aspects to be considered during the model building process (Modified from Dormann et al. (2012) *Correlation and process in species distribution models: bridging a dichotomy.* Copyright © 2012 by John Wiley Sons, Inc. Reprinted by permission of John Wiley & Sons, Inc).

| Topic | Relevant issues |
|---|---|
| Assumptions | Error structure, structure of functional relationships, relevant processes/predictors, equilibrium with environment |
| Information required | Data on distribution, populations, environments, environmental data, ecological and biological knowledge |
| Determination of model structure | Variable selection, alternative functional relationships, submodels |
| Verification | Technical correctness, model diagnostics |
| Validation | Cross-validation, external validation, parameter validation, sensitivity, specificity |
| Sources of uncertainty in model predictions | Input data, model misspecification, regression dilution, stochasticity |
| Equifinality | Over-parameterization, collinearity, non-identifiability |
| Extrapolation | Model domain, (micro-)evolution, stationarity of limiting factors and interactions, phenotypic plasticity |
| When to stop: accuracy versus complexity | Deployment time, re-parameterization, sensitivity analysis |
| Communicability and model transparency | Documentation, open source code/software |
| Knowledge potentially gleaned from the model | Surprise, emergence |
| Common errors and misuses | Lack of uncertainty analysis, use beyond purpose, overconfidence in communication |

#### 4.2.3.1.1 Indigenous and local knowledge (ILK)

Indigenous people, with collective knowledge of the land, sky and sea, are excellent observers and interpreters of changes in the environment. Their knowledge may offer valuable insights, complementing scientific data with chronological and landscape-specific precision and detail that is critical for verifying models and evaluating scenarios developed by scientists at much broader spatial and temporal scale.

Moreover, ILK provides a crucial foundation for community-based actions that sustain the resilience of social-ecological systems at the interconnected local, regional and global scales (Raygorodetsky, 2011). Indigenous and local observations and interpretations of ecological phenomena at a much finer scale have considerable temporal depth and highlight elements that may be marginal or even new to scientists.

ILK can potentially supplement other scientific data in modelling, as input to the model but also in the interpretation and understanding of the outputs of model runs. Traditional or indigenous knowledge is a result of a long series of observations transmitted from generation to generation (Berkes et al., 1995). Such 'diachronic' observations (i.e. observations over time) can be of great value and complement the 'synchronic' observations (i.e. observations made at the same time, but at different locations) that are often used for model construction and testing (Gadgil et al., 1993). Knowledge holders have not only developed a stake in conserving biodiversity, but also in understanding the complexities and interrelations among the varied entities that an ecosystem encompasses (Slobodkin, 1961). Modelling for biodiversity conservation and ecosystem services can therefore benefit significantly from the application of ILK, which may fill gaps in biodiversity modelling (Thaman et al., 2013; WWF, 2013).

ILK thus has the potential to contribute to global environmental assessments, posing the challenge of how to integrate different scales and how to connect different knowledge systems to complement each other. One of the approaches of IPBES, the 'Multiple Evidence Base approach' was developed at the Stockholm Resilience Centre as a conceptual framework for connecting diverse knowledge systems (Tengö et al., 2013).

Integration of ILK in research techniques such as modelling and remote sensing can provide a robust contribution to informed decision making. An example is animal herd management in the Arctic, where remote satellite sensing, meteorology and modelling are complemented with the indigenous knowledge of Sami and Nenets reindeer herders to co-produce datasets. The indigenous observers are able to make sense of complex changes in the environment through the qualitative assessment of many factors, complementing the quantitative assessment of variables made by scientists (Magga et al., 2011). Case studies from Canada and New Zealand also provide evidence that a combination of traditional ecological knowledge and science to understand and predict population responses can greatly assist co-management for sustainable customary wildlife harvests by indigenous peoples (Moller et al., 2004).

#### 4.2.3.2 Correlative models

Correlative models are generally easy to apply and do not require extensive knowledge of underlying processes, but instead use statistical methods to establish direct relationships between environmental variables and biodiversity data such as species richness, abundance or distribution (Morin and Lechowicz, 2008). These models produce information on biodiversity patterns and responses to drivers based on empirical observations, and do not attempt to explain the mechanisms underlying those patterns and responses (Rahbek et al., 2007). When using the correlative modelling approach, it is recognised that there are clear limitations to ecological knowledge for model development, and often the focus is on ensuring a pragmatic model implementation that will capture current existing ecological patterns, which often provides good – if narrow – projections (Araújo and Pearson, 2005; Elith and Leathwick, 2009).

Correlative models are frequently used to assess the impacts of human activities on biodiversity, forecast future impacts of environmental changes, support human productive activities (e.g. enhance agricultural production) and conservation actions (e.g. identify sites for translocations and reintroductions, or predict the location of rare and endangered species), and understand species' ecological requirements, among other uses (Peterson, 2006; de Souza Muñoz et al., 2011). Correlative models have the advantage of being tractable and easy to interpret, and permit the predictability of phenomena that depend on differences between components – for example the invasive potential of a species depends on the difference between potential and actual distributional areas (Peterson, 2006).

Correlative models can be applied at all spatial scales after careful assessment of relevant environmental predictors and response variables relevant to the question addressed (Elith and Leathwick, 2009; Guillera Arroita et al., 2015). For instance, the effect of climate variables is better assessed at large spatial extents, such as regions, and coarse resolution data may be acceptable, whereas the effect of land use or soil nutrients requires fine resolution data to cover fine-scale variations, and is usually modelled at smaller extents such as landscapes. When the selected environmental predictors act at different scales, hierarchical models with nested sub-models can be used (Elith and Leathwick, 2009). Regarding temporal scales, correlative models are often static (i.e. assume that the species-environment relationships do not change over time), and therefore often fail to capture species or community dynamics such as species dispersal. Nevertheless, temporal predictors – such as variability of food resources – may be added to models to capture variation in the state of biodiversity variables.

Correlative models should be used carefully when extrapolating biological descriptors to new spatial areas and time frames (i.e. hindcasting and forecasting applications). This is due to the possibility that conditions (e.g. climatic conditions) associated with the training data (i.e. the data used to fit the model) may not remain constant over time (Elith and Leathwick, 2009; Araújo and Peterson, 2012), or may be inadequate to represent the conditions found outside their area of distribution. Moreover, correlative models are data demanding, requiring robust datasets. However, because the data required by correlative models are often available across a range of scales, and because the models can implicitly capture many complex ecological responses, Elith et al. (2010) anticipate the continued use of correlative models for biodiversity projections.

#### 4.2.3.3 Process-based models

Process-based models are generally more complex to develop than correlative models as they require more knowledge of the processes that shape biodiversity patterns, including an explicit consideration of selecting which processes to include. These models nevertheless allow a more explicit representation than correlative approaches of ecological processes mediating biodiversity and ecosystem responses to environmental drivers. As they tend to build on a formal framework with varying levels of theoretical underpinning, they are also more capable of explaining why biodiversity patterns occur, rather than simply demonstrating that they do. The golden standard for modelling, however, frequently includes the degree to which models can be used for predictive purposes, and while this is an area in which process-based models may have an advantage over correlative models, it should also be acknowledged that the capabilities of process-based models with regard to predicting the consequences of anthropogenic impacts for biodiversity and ecosystems are uncertain and under continuous development. In response to climate change, species may change their climatic niches along three non-exclusive axes: time (e.g. phenology), space (e.g. range) and self (e.g. physiology), as described by Bellard et al. (2012). Of these, the physiological axis in particular calls for the capacity to handle evolutionary adaptations (see for more detail Section 4.3.1.1). It should also be noted that data availability generally places limits on how reliably models can be parameterized.

One example of an approach used to overcome the limitations of correlative methods is the dynamic energy budget theory (e.g. Kooijman, 2009). This is a good example of mechanistic theory that aims to capture the quantitative aspects of metabolism at the organism level from a small set of key assumptions (Sousa et al., 2008). The dynamic energy budget theory makes it possible to account for the effects of environmental variability on organisms through food and temperature changes and captures the diversity of all possible living forms on Earth in a single mechanistic framework. This allows the representation of the energetics and major life history traits of all possible species in a community with the same set of unspecific taxa-dependent dynamic energy budget parameters.

Overall, process-based models are limited by the number of processes that are explicitly included, the sensitivity of the system dynamics to the mathematical form used to represent the process, the sensitivity to the data used to estimate the parameters, and the limited capacity to predict beyond the range of observed conditions. Despite the wide use of process-based ecosystem models in biology and ecology they, as do all other model developments, suffer from fundamental and practical limitations.

Various strategies and approaches for process formalisation can be distinguished among the available process-based models:

***Box models***. This is the simplest and most developed category. It describes ecosystem dynamics using a set of state variables (e.g. fish biomass) that are connected together by fluxes (e.g. consumption or predation) based on given functional responses that are either predefined (Holling, 1959) or emergent properties (Ahrens et al., 2012). The most common use of this type of model is to simulate mass balances and energy fluxes at the scale of the system represented, and this is one of its main advantages. On the other hand, they tend to use highly aggregated representations of state variables (e.g. lumping all fish species at a trophic level together) and therefore neglect phenomena such as the importance of size in controlling metabolism, predator-prey interactions and life history omnivory (i.e. dietary changes as organisms grow).

***Age/stage/size-structured models***. These models are box models that are structured along a dimension that is assumed to be functionally important. They explicitly account for some processes of metabolism such as growth, reproduction and the age-dependence of respiration. Age/stage-structured models are widely used for fisheries management (see Hilborn and Walters, 1992), as well as for food web models (e.g. Walters et al., 2010). Size-structured models emphasise the impact of size as a structuring element in ecosystems. In marine and freshwater ecosystems, size is usually a good predictor of trophic level at the community level (Jennings et al., 2001) because many predators are size-selective, leading to this biological trait to exert a strong influence on predation and metabolism. Size-based models are easier to parameterise than functional group or age/stage-structured food-web models, though in particular applications there may be more interest for species than for size *per se*. Size-structured models can, however, be constructed with explicit species considerations to make them more suitable for addressing questions of direct relevance to biodiversity research (Shin and Cury, 2001; Blanchard et al., 2014).

#### 4.2.3.4 Hybrid models: combining correlative and process-based modelling

Hybrid models combine correlative and process-based modelling approaches (Schurr et al., 2012) in order to represent complex, integrated systems with a focus on biophysical as well as human components (Parrott, 2011). Such models tend to be highly data-driven and help build on our understanding of important factors and synthesise knowledge, as well as providing a structural link between data sources and decision-support systems. Hybrid model development takes a pathway in which some of the ecological processes defining the ecological system under study (e.g. the realised niche) are modelled explicitly (i.e. process-based), while others are based on correlative niche modelling (Thuiller et al., 2013). Hybrid approaches derive from the interest to balance realism and flexibility in model building with limited knowledge, but this approach also comes with important challenges.

How different models are integrated into hybrid approaches is often a difficult issue. Gallien et al. (2010) indicate that one of the current limitations of the hybrid approach is the form and strength of the relationship between habitat suitability and demographic parameters. Changes in habitat suitability are normally integrated with population processes by limiting carrying capacity. Furthermore, the response of ecological processes (e.g. growth, dispersal and thermal tolerance) to environmental changes is unclear, and is often assumed to be unimodal or linear. Non-linear functional response could make the model more complex.

Broadly speaking, mechanisms determining ecosystem dynamics can be related to the ecology of species, species interactions and evolutionary processes (Lavergne et al., 2010). Any biological process of interest should have an explicit link with the components formulated in the model. However, this link does not need to be one-on-one (Lurgi et al., 2015). The implementation of these processes in the model may be carried out in a wide variety of ways spanning a broad range of complexities, from cellular automata (Iverson and Prasad, 2001), meta-population models (e.g. Wilson et al., 2009) and structured meta-population models (Akçakaya et al., 2004), to spatially-explicit population models (e.g. Cabral and Schurr, 2010), individual-based models (e.g. Grimm et al., 2005), trophic models (e.g. Albouy et al., 2014) and reaction-diffusion models (e.g. Wikle, 2003; Hui et al., 2010). For example, the recently introduced 'dynamic range modelling' framework (Pagel and Schurr, 2012), based on a Bayesian approach, overcomes several of these limitations as it uses species distribution data and time series of species abundance to statistically estimate both distribution dynamics and the underlying response of demographic rates to the environment. This approach is particularly relevant when dispersal limitation or source-sink dynamics cause disequilibrium between species distributions and environmental conditions (Pulliam, 2000).

The dynamic bioclimate envelope model developed by Cheung et al. (2008b) simulates changes in the relative abundance of marine species through changes in population growth, mortality, larval dispersal and adult movement following the shifting of the bioclimate envelopes induced by changes in climatic variables. The model does not account for species interactions and potential food web changes, which are however considered in a combined food web and habitat capacity model (Christensen et al., 2014). Dynamic bioclimatic envelope models are also being developed to account for effects of ocean biogeochemistry, such as oxygen level and pH, on the eco-physiology and distribution of marine fish (Stock et al., 2011). Models with emergent dynamics may also include species interactions (e.g.Albert et al., 2008) or abiotic processes included via feedbacks (e.g. wildfires versuss vegetation growth; Grigulis et al., 2005).

# 4.3 Available approaches to modelling the impact of drivers on biodiversity and ecosystem functioning

## 4.3.1 Modelling approaches addressing biological levels of particular relevance to IPBES

### 4.3.1.1 Individual-level models and evolutionary adaptation

Populations are not static, but evolve. As a consequence, species may be able to adapt to conditions different from those previously experienced (Hoffmann and Sgrò, 2011). As introduced in Figure 4.2, evolution can alter dispersal patterns, physiology and biotic interactions (Thuiller et al., 2013), and this poses a clear problem for predictive modelling at all levels, from genes to ecosystems: how to make predictions that go beyond current conditions?

There has been considerable research aimed at addressing this question, notably theoretical models that explicitly account for biological processes such as mutation, dispersal and interactions within and between species (e.g. mating and competition) (Bürger, 2000). Such models can account for environmental change and allow projections about future scenarios, beyond the range of what is currently observed. They also provide a means of assessing the robustness of predictions across uncertain parameters and processes.

Short-term evolutionary projections focus on the response to selection within a population based on the initial ('standing') genetic variance, and can account for selection acting on multiple traits (Lande and Arnold, 1983). Assuming that several genes underlie these traits, quantitative genetic models can accurately predict short-term evolutionary responses to a changing environment, given information about the genetic variance for each trait, the covariance among traits, and the strength of selection induced on each trait (see, for example, Shaw and Etterson, 2012). In practice, this information is unavailable for most species and over large spatial extents. Thus, ranges of plausible values must be inferred – with uncertainty – based on data from other species.

Longer-term projections are made difficult by the need to account for the dynamics of genetic variation. Processes such as mutation and migration that build genetic variance must be modelled (Barton and Turelli, 1989). Selection itself causes allele frequency changes that can increase or decrease genetic variance (de Vladar and Barton, 2014).

While many of these models assume a stable population size, more relevant to our understanding of biodiversity change are models that explicitly account for the feedback between population dynamics and evolutionary change. One theoretical approach focuses on key ecological traits (e.g. resource acquisition traits) that impact population dynamics and whose optimum values shift in a changing environment (Pease et al., 2008; Duputié et al., 2012). Such models that account for population dynamics are essential for addressing the extinction risk faced by a population. How far and how fast can a population be pushed by environmental change before it collapses (Bürger and Lynch, 1995; Lande and Shannon, 1996; Gomulkiewicz and Houle, 2009)? These models identify the critical speed of environmental change above which evolutionary lags grow over time until populations can no longer persist.

While the above models focus on standing genetic variance, some environmental changes require novel genetic solutions. Recent models have asked when new mutations can 'rescue' a population before it goes extinct following an environmental perturbation (e.g. Bell and Collins, 2008; Bell, 2013). These models provide key insights into the factors that promote evolutionary rescue, including the population size, the severity of environmental degradation, and the array of possible rescue mutations (Carlson et al., 2014). Results from these combined evolutionary and population dynamic models can be counterintuitive. For example, while evolutionary adaptation generally works best when the environment changes slowly, evolutionary rescue can be more likely when an environmental shift occurs rapidly, because the release from density-dependent competition helps establish rescue mutations (Uecker et al., 2014).

While the simplest evolutionary models are not spatially explicit, models are increasingly examining how the arrangement of populations and migration rates among them influence evolutionary processes in the face of a changing environment. For example, models have explored the process of evolution to a new or altered environment in the face of migration from the rest of the species range (Gomulkiewicz et al., 1999). Such models can inform policy decisions about the maintenance of gene flow and the importance of migration corridors. Other models explore how the geographical range of a species evolves over time in the face of environmental change. Interestingly, these models are highly sensitive to assumptions made about the dynamics of genetic variance and whether it is held fixed, allowed to evolve deterministically, or subjected to random genetic drift (Polechová et al., 2009, Polechová and Barton, 2015). The latter paper clarifies how demographic and evolutionary processes combine to predict whether a species will persist or undergo range contraction when the environment varies over space.

Many evolutionary models focus on genetic changes within a single species. Clearly, it is useful to clarify what might happen in simplified scenarios before adding the complexity of species interactions. To fully account for evolution in climate change models, however, we need to account for interactions among species co-occurring within a community. Those models that have considered species interactions suggest that evolutionary responses to environmental change can be fundamentally altered. For example, interspecific competition can hinder evolutionary adaptation and drive extinct a species that would be able to persist if it were on its own (Johansson, 2008). Other models demonstrate that accurate predictions require an understanding of how selection is shaped by both species interactions and environmental change (Osmond and de Mazancourt, 2013; Mellard et al., 2015).

The results of any model, particularly evolutionary models, are sensitive to the details assumed. What are the selection processes and life strategies? How far do individuals migrate? How patchy is the environment? Which mutations are neutral or functional? These details matter when predicting whether a species will persist or become extinct.

Evolutionary processes thus raise a great deal of uncertainty in our projections of future biodiversity change in the face of major environmental drivers. Models such as those described above allow us to explore the range of possibilities. Not accounting for evolutionary change is, in most cases, the most conservative assumption for the maintenance of biodiversity (Shaw and Etterson, 2012). On the other hand, allowing evolutionary change under generous assumptions about current and future levels of genetic variance allows us to delimit the most optimistic scenarios for biodiversity in the face of human-caused environmental change.

#### 4.3.1.2 Species- or population-level models

Populations are groups of organisms, all of the same species, that live in a given area and interact. Biodiversity change at the species or population level is often measured using data on population demography and species distribution (i.e. the distribution of populations within a species). Populations change in size and distribution due to the interaction between internal (e.g. growth rate, reproduction) and external (e.g. resources, predation, diseases) factors. Models building from the simple exponential function, including the logistic population model, life table matrix modelling, the Lotka-Volterra models of community ecology, meta-population theory, and the equilibrium model of island biogeography and many variations thereof, are the basis for ecological population modelling to predict changes over time (Gotelli, 2008).

Without the influence of external factors (thus in a density-independent situation), population growth can be modelled as exponential (Vandermeer and Goldberg, 2004). However, as the population size increases, density-dependence factors – such as resource limitation, competition or disease – frequently impact population growth because births and deaths are dependent on population size. Under density-dependence, growth rates slow down and reach a maximum, depicting a sigmoid curve of population size against time, in other words logistic growth. In the logistic model, the maximum number of individuals in the population is based on the carrying capacity of the system.

The logistic model is frequently used to study the impact of harvesting a population by removing individuals from it (Giordano et al., 2003). Important modifications to the original model include the introduction of critical threshold densities, fluctuations in the carrying capacity and discrete population growth. A popular, but also much debated, example of the logistic growth model is the application to managing fisheries by finding the optimal strategy that maximises the population growth rate and the long-term yields achieving the maximum sustained yield (Gotelli, 2008). Discussions around this concept are large and include the importance of including species interactions to calculate this reference point in the context of fisheries management (Walters et al., 2005).

Because species do not occur in isolation, the dynamics of any one species affects the dynamics of other sympatric species. In these cases, the logistic equation can be modified to consider the interaction of a population with interspecific competitors, with predators and with prey (Otto and Day, 2007). Lotka and Volterra models for interspecific competition and prey-predator interactions are the classical initial frameworks for competition and predation studies in ecology. These models build from the logistic equations and incorporate the interactions with other populations of competitors, predators and prey, modifying population growth rates. A classic example of the predator-prey interactions Lotka-Volterra model is the prediction of the regular cycling of the population size of Canada lynx (*Lynx canadensis*) and the snowshoe hare (*Lepus americanus*) (Sinclair and Gosline, 1997). An important concept in predator-prey interactions is the functional response of the predator as a function of the prey abundance. This response can be represented as a linear function of prey abundance (called the Type I response). More realistic assumptions incorporate handling time, under which the response of the predator increases to a maximum prey consumption rate (Type II response). A variation of the latter incorporates switching with an acceleration of the feeding rate at intermediate prey density and a decrease at high prey density as an asymptote is reached (Type III; Holling, 1959). These responses are key elements when modelling the ability of predator species to control prey populations (Gotelli, 2008).

Additionally, populations are often not closed, so that individuals tend to move between populations, influencing their persistence and survival. Different ways to model sets of populations (or meta-populations) exist. This approach is applied to study linkages of populations at the landscape scale, both in terrestrial and aquatic systems. Methodologies quantify the fraction of all population sites that are occupied, and have been notably applied to study the impacts of protected areas to inform biodiversity conservation (Royle and Dorazio, 2008; Kritzer and Sale, 2010). In addition, the number of species interacting in a specific place depends on the area available for those species to survive and the relationship between species and area holds in most assemblages of organisms worldwide.

This is at the origin of island biogeography that states that the larger an island, the more species it will hold, and the more potential interactions there will be. The original explanation for this pattern was related to habitat types, considering that larger islands include a higher diversity of habitats, and thus species restricted to those habitat types will only occur on larger islands (Gotelli, 2008). However, an alternative hypothesis developed with the equilibrium model of island biogeography includes the immigration of new species and the extinction of resident species as the main force behind the relationship between area, habitat heterogeneity and the number of species in a community (Simberloff, 1976; Allouche et al., 2012).

When survival and fecundity rates depend on the age of individuals affecting population growth, age-structured models using the analysis of life table matrices are applied (Otto and Day, 2007; Gotelli, 2008). However, many other parameters can affect vital rates and their variability in space and time, which is at the core of estimating the risk of extinction or decline of a population.

Population viability analysis, a form of risk assessment analysis, estimates these risks by identifying major threats faced by a population and by evaluating the likelihood of future population persistence (Beissinger and McCullough, 2002; Morris and Doak, 2002).

Population viability analyses are often applied to the conservation and management of threatened or rare species (Akçakaya et al., 2004), with the aim to evaluate options for how to improve the chance of survival of populations or species at risk (Akçakaya and Sjögren-Gulve, 2000; Drechsler and Burgman, 2004).

Species occurrence and abundance are often modelled using correlative methods generally described as species distribution models. Species distribution models are mainly used to evaluate 1) overall species distributions; 2) historic, present and future probability of occurrence; and 3) to gain an understanding of ecological niche limits, which is why this approach is also called ecological niche modelling (Aguirre-Gutiérrez et al., 2013).

Species distribution models are widely used to model the effects of environmental changes on species distribution across all realms (Pearson and Dawson, 2003; Brotons, 2014). The multiple applications of species distribution models are reflected in the diversity of designations used to refer to this type of modelling approach, including ecological niche models, bioclimatic envelope models, and habitat (suitability) models (Elith and Leathwick, 2009). Modelling approaches that incorporate species abundance data along with species distribution data, for a joint prediction of the effects of environmental drivers on population demography and consequently on the overall species distribution, are also being pursued (Ehrlén and Morris, 2015).

Research that incorporates expert knowledge into species distribution models is relatively limited. However, in a study on species distribution modelling, Niamir et al. (2011) incorporated existing knowledge into a Bayesian expert system to estimate the probability of a bird species being recorded at a finer resolution than the original atlas data. They noted that knowledge-based species distribution maps produced at a finer scale using a hybrid model/expert system had a higher discriminative capacity than conventional approaches, even though such an approach might be limited to well-known species. Furthermore, in a study to evaluate trade-offs for using species occurrence data in conservation planning, Rondinini et al. (2006) noted that the geographic range data of species generated by expert knowledge had the advantage of avoiding the potential propagation of errors through data processing steps.

#### 4.3.1.3 Community-level models

Community-level modelling offers an opportunity to move beyond species-level predictions and to predict broader impacts of environmental changes (e.g. Hilbert and Ostendorf, 2001; Peppler-Lisbach and Schröder, 2004; D'Amen et al., 2015), which may be relevant in certain decision-making contexts. For example, it can be used to predict the impact of losing a top predator in the structure of a trophic network or the impacts of land-use change in native communities. Community-level approaches are also recommended when time and financial resources are limited, when existing data are spatially sparse or when the knowledge on individual species distribution is limited (Ferrier et al., 2002a) or even absent, as in the case of non-described species in highly diverse environments, and when species diversity is beyond what can feasibly be modelled at the individual species level. Overall, assessing changes in community composition, including both species presence and abundance, and how those changes affect ecosystem processes, provides a more detailed understanding of the impacts of drivers (Newbold et al., 2015). Moreover, species richness – a community-level metric – is a commonly used biodiversity indicator (Mace et al., 2012).

Community-level distribution models, as for species distribution models, use environmental data to predict the distribution of species assemblages or communities. Data input needs are similar to species distribution model inputs but model outputs are more diverse and can be classified into five main types (Ferrier and Guisan, 2006): community types (groups of locations with similar species composition), species groups (groups of species with similar distributions), axes or gradients of compositional variation (reduced space dimensions of compositional patterns), levels of compositional dissimilarity between pairs of locations, and various macro-ecological properties (e.g. species richness) and even phylogenetic diversity.

Ferrier and Guisan (2006) and D'Amen et al. (2015) identify three approaches to community-level modelling (Figure 4.4): 1) 'assemble first, predict later', whereby species data are first combined with classification or ordination methods and the resulting assemblages are then modelled using machine learning or regression-based approaches, 2) 'predict first, assemble later', whereby individual species distributions are modelled first and the resulting potential species distributions are then combined (i.e. the result is in fact the summation of individualistic models), and 3) 'assemble and predict together', whereby distributions of multiple species are modelled simultaneously using both environmental predictors and information on species co-occurrence patterns.

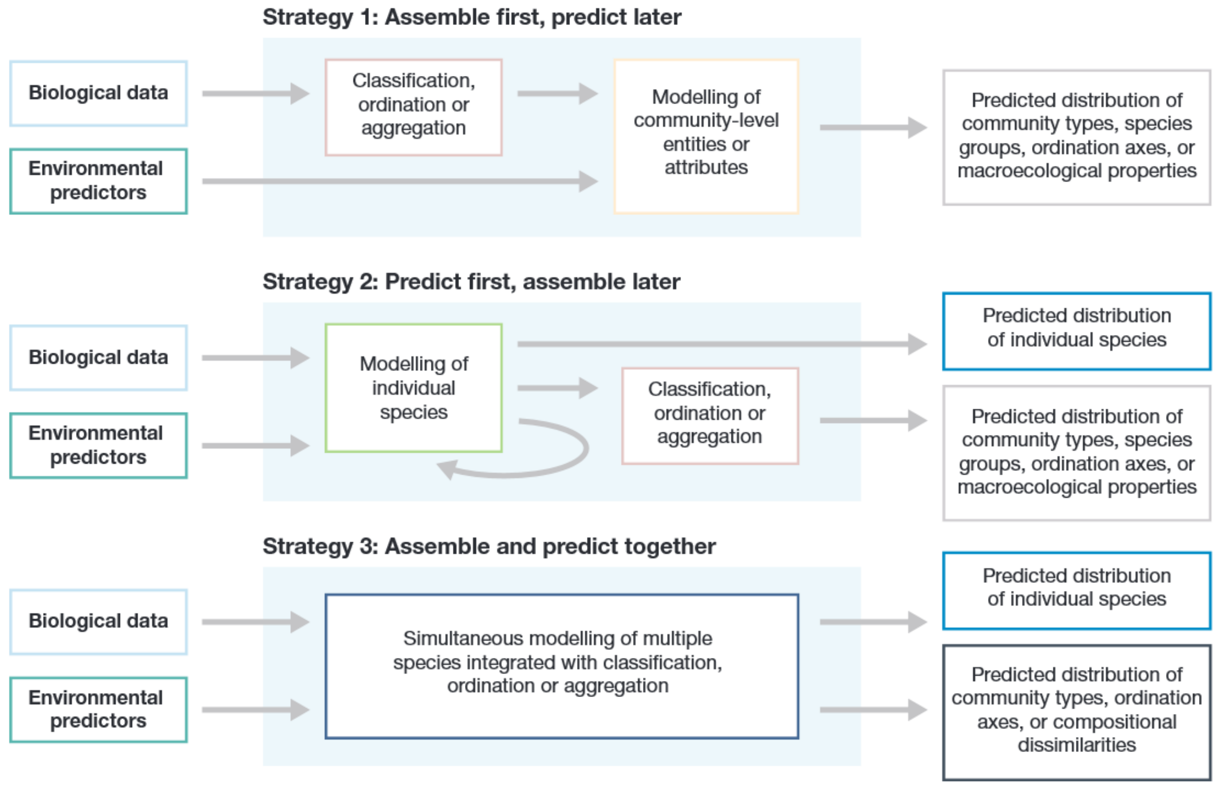


**Figure 4.4**: Main approaches to community-level distribution models (Modified from Ferrier and Guisan, 2006. *Spatial modelling of biodiversity at the community level.* Copyright © 2006 by John Wiley Sons, Inc. Reprinted by permission of John Wiley & Sons, Inc).

These approaches have different strengths (D'Amen et al., 2015). The first and third approach are more able to capture overall patterns of response and are better options if rare species, for which distribution data may be scarce, represent a significant fraction of the species assemblage. However, the second approach allows more flexibility in how different species respond to different environmental factors, though it may fail to produce reliable projections of rare species distributions (Ferrier and Guisan, 2006). Similar reasoning can be used when deciding whether to use species distribution models or community-level models to assess community responses. Species distribution models can provide more reliable predictions of well-sampled species, but may fail with rare species and are resource-demanding when applied at the community level.

The strengths and weaknesses of community-level modelling approaches and the applicability of community models are discussed by Ferrier et al. (2002b) and by Ferrier and Guisan (2006). More recently, D'Amen et al. (2015) have highlighted potential research avenues and proposed novel integrative frameworks to encourage the state-of-the-art in spatial predictions at the community level. As in species distribution models, correlative community-level distribution models can also integrate ecological processes such as meta-community dynamics and species interactions (Mokany and Ferrier, 2011) to enhance their predictive ability (D'Amen et al., 2015).

#### 4.3.1.4 Ecological interaction networks

Ecological interaction networks include, among other examples, trophic webs and plant-pollinator webs (Ings et al., 2009). Species interactions within communities can be explicitly modelled using process-based approaches that describe the links between species and the dynamics that determine species coexistence in the network, such as predator-prey oscillations (Verhoef and Olff, 2010).

Network topology is also an important consideration when building interaction models, since the links between elements may follow a non-random pattern. In food webs, interactions patterns are shaped by body size, which justifies the use of size-structured models (Woodward et al., 2005; Loreau, 2010).

Correlative approaches are also frequent in studies of interaction networks, due to their lower information requirements, but Ings et al. (2009) advocate against the use of inferential approaches and recommend pursuing more mechanistic approaches that build on first principles and ecological theory. Similarly, applications in modelling marine ecosystems will require the coupling of different trophic levels that may have different characterisations. One way to represent biodiversity in complex marine systems would be to concentrate the detail of representation at the target species level and their main interactions at the community level (FAO, 2008). Community interaction network approaches have been used to assess the impacts of, for example, invasive species (Woodward and Hildrew, 2001), the overfishing of top predators (e.g. Bascompte et al., 2005), biodiversity and ecosystem function relationships (Fung et al., 2015), freshwater pollution (e.g. Scheffer et al., 1993) and global warming (Petchey et al., 1999).

Outputs from community-level distribution models can be used to inform species traits approaches, assessing the composition of impacted communities. Species traits approaches can also be linked to interaction network models to predict how changes in community traits will affect ecosystem functioning (Harfoot et al., 2014b). Species traits approaches move the focus from species composition in a community to the distribution of traits or average trait values in the community. Species traits underlie species responses to drivers, that is, their ability to cope with environmental change, but also their role in environmental processes. Therefore, the distribution of trait values in a community (e.g. root depth, body size or forage range) may not only inform on the vulnerability of the community to changes in drivers, but also on the effects of community compositional change to ecosystem functioning, and consequently to ecosystem services (Lavorel and Garnier, 2002; Suding et al., 2008; Oliver et al., 2015). Trait-based ecological risk assessment is an example of a trait-based approach to assess ecological responses to natural and anthropogenic stressors based on species characteristics related to their functional roles in ecosystems (Baird et al., 2008).

Another approach commonly used to assess community change over time is through species-area relationship models. These are used to predict species richness as a function of habitat area. Species-area relationship models have been tested and applied to a wide range of taxa and across all scales, from local to global (e.g. Brooks et al., 2002; Brooks et al., 1997). Species-area relationship models are often used to predict the impacts of changes in habitat availability, driven by land-use change (e.g. van Vuuren et al., 2006; Desrochers and Kerr, 2011) or climate change (e.g. Malcolm et al., 2006; van Vuuren et al., 2006), on community richness, but also to assess the impacts of direct exploitation on community parameters such as species turnover rates (e.g. Tittensor et al., 2007). Reviews on the use of species-area relationships can be found in Rosenzweig (2010), Drakare et al. (2006) and Triantis et al. (2012).

The most common species-area relationship model is the power function (Arrhenius, 1921), $S=cA^z$, where S is species richness, A is habitat area, and c and z are model parameters (Rosenzweig, 2010).

Notwithstanding the general use of the power function, species-area relationship models may be best described by other functions or by averaging the predictions of alternative models (i.e. multi-model species-area relationship approaches) when there is no single best model (Guilhaumon et al., 2008). Another important caveat relates to the risk that species-area relationship models may overestimate predicted species loss due to habitat loss (Pereira and Daily, 2006). This limitation can be addressed through the use of modified species-area relationship approaches that better represent community dynamics, such as the species-fragmented area relationship (Hanski et al., 2013) – which considers the effects of habitat fragmentation on species diversity patterns – and the countryside species-area relationship (Proença and Pereira, 2013) – which accounts for the differential use of habitats in a landscape by different species groups.

#### 4.3.1.5 Ecosystem-level models and integrated models

Ecosystem-level models may focus on the biophysical dimension of ecosystems (e.g. dynamic global vegetation models), or they can be developed to also include economic and social aspects (e.g. EwE models, see Chapter 5).

Dynamic Global Vegetation Models (DGVMs) are process-based models that simulate various biogeochemical, biogeophysical and hydrological processes such as photosynthesis, heterotrophic respiration, autotrophic respiration, evaporation, transpiration and decomposition.

DGVMs are the most advanced tool for estimating the impact of climate change on vegetation dynamics at the global scale (Smith et al., 2001). They simulate shifts in potential vegetation and the associated biogeochemical and hydrological cycles as a response to shifts in climate. DGVMs use time series of climate data and, given the constraints of latitude, topography and soil characteristics, simulate monthly or daily dynamics of ecosystem processes. DGVMs are most often used to simulate the effects of future climate change on natural vegetation and carbon and water cycles, and are increasingly being coupled with atmosphere-ocean general circulation models to form Earth system models.

The basic structure of a DGVM is shown in Figure 4.5.

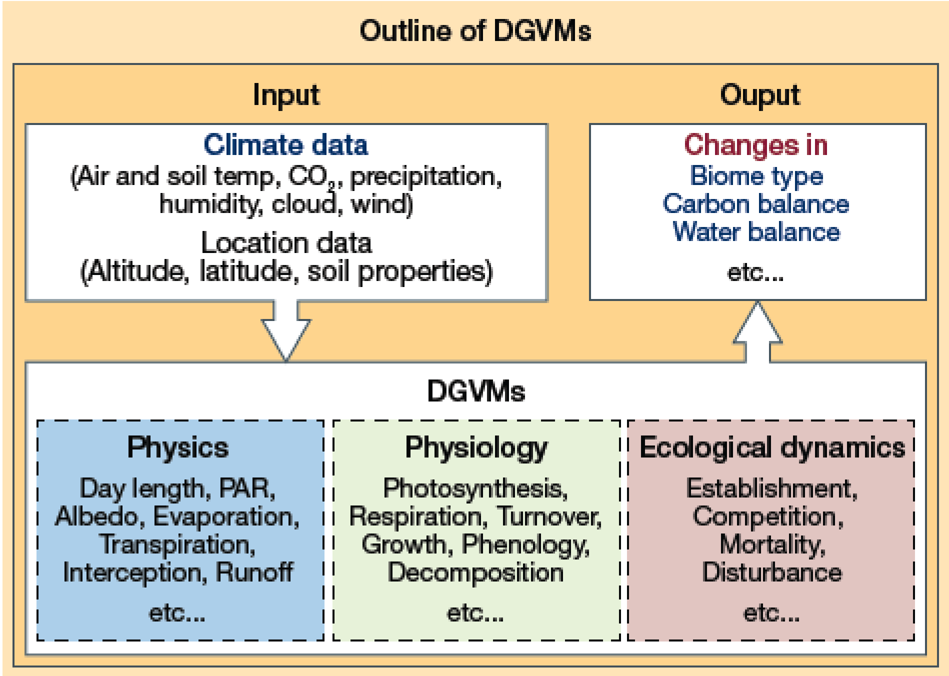


**Figure 4.5**: Structure of Dynamic Global Vegetation Models (Modified from: http://seib-dgvm.com/oview.html).

DGVMs capture the transient response of vegetation to a changing environment using an explicit representation of key ecological processes such as establishment, tree growth, competition, death and nutrient cycling (Shugart, 1984; Botkin, 1993). Plant functional types are central to DGVMs as, on the one hand, they are assigned different parameterisations with respect to ecosystem processes (e.g. phenology, leaf thickness, minimum stomatal conductance, photosynthetic pathway, allocation and rooting depth) while, on the other hand, the proportion of different plant functional types at any point in time and space defines the structural characteristics of the vegetation (Woodward and Cramer, 1996).

The key advantages of using DGVMs include the capacity to simultaneously model the transient responses related to dynamics of plant growth, competition and, in a few cases, migration. As such, this allows the identification of future trends in ecosystem functioning and structure and these models can be used to explore feedbacks between biosphere and atmospheric processes (Bellard et al., 2012). DGVMs are, however, focused on a limited number of plant functional types, which induces a high level of abstractedness (Thuiller et al., 2013).

Adding a further level of complexity beyond ecosystem modelling is achieved through integrated assessment models (IAMs, see Figure 4.6), which were defined in the IPCC Third Assessment Report (IPCC, 2001) as 'an interdisciplinary process that combines, interprets, and communicates knowledge from diverse scientific disciplines from the natural and social sciences to investigate and understand causal relationships within and between complicated systems'.

It is generally agreed that there are two main principles to integrated assessment: integration over a range of relevant disciplines, and the provision of information suitable for decision making (Harremoes and Turner, 2001). IAMs therefore aim to describe the complex relationships between environmental, social and economic drivers that determine current and future states of the system and the effects of climate change, in order to derive policy-relevant insights (van Vuuren et al., 2009). One of the essential characteristics of integrated assessment is the simultaneous consideration of the multiple dimensions of environmental problems. At the global level**,** IAMs could potentially be a valuable tool for modelling biodiversity dynamics under different drivers; however, current IAMs are not developed for this application (Harfoot et al., 2014a). Existing IAMs are largely used for modelling climate change and investigating options for climate mitigation. Key outputs from IAMs include anthropogenic greenhouse gas emissions. However, these also provide projections for other variables, such as land cover and land use (including deforestation rates).

One of the most noticeable limitations of IAMs is that they focus largely on terrestrial systems, not marine or freshwater aquatic ecosystems (as shown in Figure 4.6, which provides a schematic representation of a typical IAM). Another notable limitation is the lack of feedback from changes in biodiversity, ecosystem functions and terrestrial ecology on other drivers such as climate change and land-use change. For example, actions that reduce the number or composition of species in natural systems may compromise ecosystem functioning, as the ability of ecosystems to provide services may depend on both these aspects (Tilman et al., 2001; Loreau et al., 2001; Hooper et al., 2005; Isbell et al., 2011). At the European level, CLIMSAVE not only integrates sectoral models, but also has feedbacks and can be used to explore the impacts of selected adaptation options (Harrison et al., 2015).

IAMs typically describe the cause-effect chain from economic activities and emissions to changes in climate and related impacts on, for example, ecosystems, human health and agriculture, including some of the feedbacks between these elements. To make their construction and use tractable, many IAMs use relatively simple equations to capture relevant phenomena, for example for the climate system and carbon cycle (Goodess et al., 2003). However, the behaviour of these components can have a significant impact on IAM results and the quality of policy advice, with the possibility of simplifications in the Earth system projections leading to imprecision (or even error) in projecting impacts and costs of mitigation.

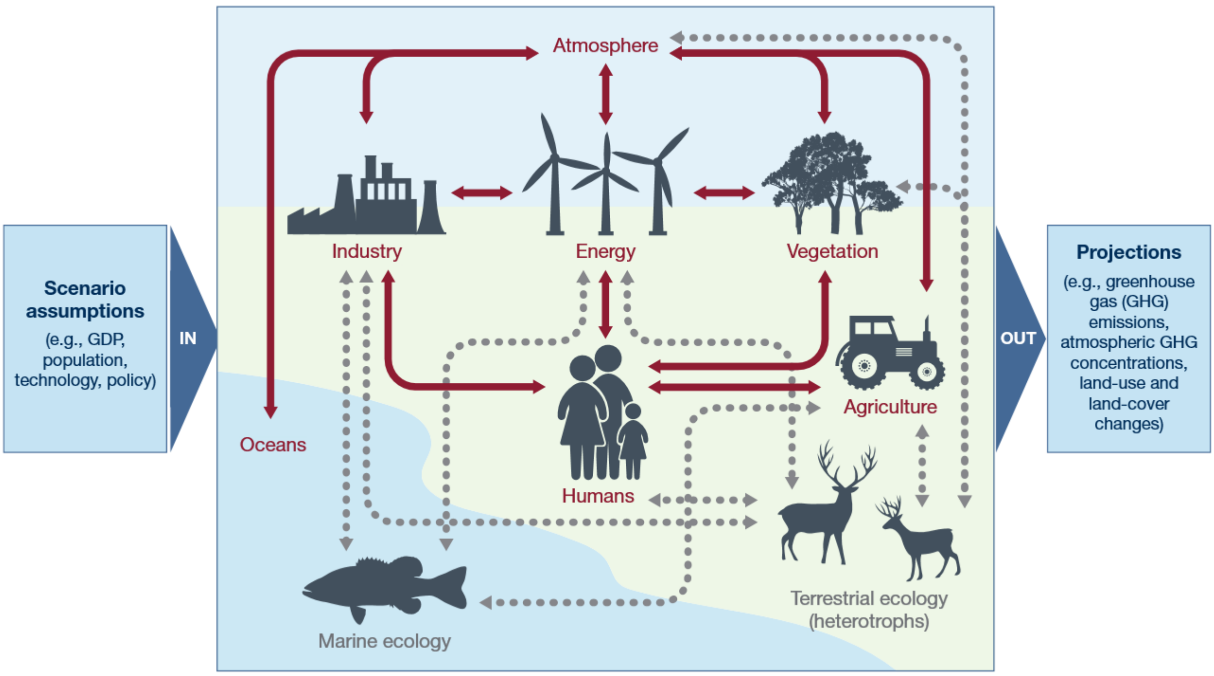


**Figure 4.6:** Schematic representation of a typical full-scale integrated assessment model. Red labels and arrows represent existing model components and interactions, while grey labels and greydashed arrows indicate

important components and interactions not currently included (Modified from Harfoot et al., 2014a. *Integrated assessment models for ecologists: the present and the future.* Copyright © 2014 by John Wiley Sons, Inc. Reprinted by permission of John Wiley & Sons, Inc).

Over the last decade, IAMs have expanded their coverage in terms of land use and terrestrial carbon cycle representation, non-$CO_2$ gases and air pollutants, and by considering specific impacts of climate change. Some IAMs have a stronger focus on economics, such as multi-sectoral computable general equilibrium models that are combined with climate modules and models focused on cost-benefit analysis; others focus on physical processes in both the natural system and the economy (integrated structural models/biophysical impact models). Examples of IAMs are IMAGE (Integrated Model to Assess the Global Environment), DICE (Dynamic Integrated model of Climate and the Economy), FUND (Climate Framework for Uncertainty, Negotiation and Distribution) and MERGE. All of these models include key drivers of change such as population and macro-economy that can be derived from various external and internal sources.

However, as IAMs aim to integrate different aspects of the environment, they run the risk of becoming extremely complex. The developers of such models therefore have to make decisions about the focus of their study and how to express the impacts they estimate, whether it is through the reporting of physical changes in emissions, shifts in land-use activity or mortality rates, or through cost-benefit analyses of damages resulting from climate change (Goodess et al., 2003). The data requirements for these IAMs are also large and not always feasible.

## 4.3.2 Modelling options, strengths and limitations

### 4.3.2.1 Meeting policy information needs

Models allowing the assessment of impacts of changes in drivers on biodiversity or ecosystem processes are important tools to support decision making. To be effective, models should be able to address the policy or decision-making needs that motivate their use. A formal and accurate definition of the decision-making context is therefore essential in this process (Guisan et al., 2013). A precise definition of the policy or decision context should inform the selection of modelling framework, including model complexity, spatial and temporal scales or response variables and data requirements (Chapter 2). State variables should be sensitive to the pressures underlying alternative management scenarios or addressed by policies and, if possible, be responsive at temporal and spatial scales that are relevant for policy strategies. For example, small farmland birds are responsive to agro-environmental schemes implemented at the field scale, while large farmland birds are more affected by activities over larger spatial scales (Concepción and Díaz, 2011). Moreover, state variables should also be representative of the biodiversity attributes underpinning the benefits of nature that are valued in a given decision-making context.

Regarding model scope, models should be adjusted to the specific requirements of the decision-making context. Models could rely on observed data to describe the relationship between pressures and response variables, explicitly describe the processes linking those variables, or follow an intermediate approach. The explicit inclusion of mechanisms in modelling approaches will be relevant whenever the understanding of the underlying dynamics is necessary to guide management and where changing environmental conditions call for a mechanistic approach (Gustafson, 2013; Collie et al., 2014). The use of correlative approaches, on the other hand, is suitable where there is limited knowledge about the underlying mechanisms or when model outputs are able to capture the dominant response patterns that are needed to inform policy, such as the evaluation of large-scale conservation initiatives (Araújo et al., 2011; Dormann et al., 2012).

As for model complexity, input data requirements should be balanced against data availability and quality – namely the spatial and temporal resolution of available data – as a lack of adequate input data may compromise model feasibility and the quality of results (Collie et al., 2014). The ongoing development of new technologies and remote sensing to monitor species and ecosystems, as well as platforms for data sharing, is encouraging as it is resulting in increased data availability and accessibility (Pimm et al., 2014). The integration of local observations and remote sensing products can provide a more complete view of the responses of biodiversity to environmental change and can improve the modelling of ecosystem processes across scales (Pereira et al., 2013; Pimm et al., 2014).

**Table 4.3:** Summary of major biodiversity models and modelling approaches.

| Model | Level of organization | Model type | Level of integration | Required level of expertise | Examples | References |
|---|---|---|---|---|---|---|
| Evolutionary models | Organisms | Mixed (hybrid) | Integrated models | High | How demographic and evolutionary processes combine to predict whether a species will persist or not | Polechová and Barton, 2015; Barton, 2001 |
| Dynamic Energy Budget models | Organisms | Mechanistic | Integrated models | High | To understand evolution of metabolic organisation | Kooijman, 2009 |
| Aquatic habitat suitability | Community | Expert-based models | Single model | Basic | To link environmental conditions to the quantitative habitat suitability of aquatic species | Mouton et al., 2009 |
| Species Distribution Models | Species/ Populations | Mainly correlative | Single or integrated models | Basic – Moderate | Used to model the effects of environmental change on species distribution | Pearson and Dawson, 2003; Elith and Leathwick, 2009; Stockwell and Peters, 1999; Phillips et al., 2006 |
| Dynamic bioclimate envelope model | Species/ Populations | Mixed (hybrid) | Integrated models | Moderate | Changes in the relative abundance of marine species induced by change in climatic variables | Cheung et al., 2008a, 2008b, 2011; Gallego-Sala, 2010; Notaro et al., 2012; Fernandes et al., 2013 |
| Age/stage-structured models | Species/ Populations | Correlative | Single model | Basic | Widely used for fisheries management | Hilborn and Walters, 1992; Getz, 1988; Barfield et al., 2011 |
| Food web models | Ecosystems | Process-based | Integrated models | Moderate | Widely used for ecosystem-based management | Christensen and Walters, 2011 |
| Size-based models | Community | Correlative | Single model | Basic | Impact of size in marine and freshwater ecosystems management | Duplisea et al., 2002; Rochet et al., 2011 |
| Species-Area Relationship models | Community | Correlative | Single model | Moderate | Used to predict the impacts of changes in habitat availability, driven by land use change or climate change | van Vuuren et al., 2006; Desrochers and Kerr, 2011; Pereira et al., 2013; Huth and Possingham, 2011 |
| Biodiversity metric models | Community | Correlative | Integrated models | Moderate | A quantitative and integrated approach to assess the biodiversity with multiple indicators | Janse et al., 2015 |
| Lotka-Volterra | Community | Process-based | Integrated models | High | For interspecific competition and prey-predator interactions | Sinclair and Gosline, 1997 |
| Dynamic Global Vegetation Models | Ecosystem | Process-based | Integrated models | High | To estimate the impact of climate change on vegetation dynamics at global scale and its carbon and water cycles | Botkin, 1993; Bellard et al., 2012; Cramer et al., 2001 |
| General ecosystem model | Global | Process-based | Integrated models | High | Uses a unified set of fundamental ecological concepts and processes for any ecosystem to which it is applied, either terrestrial or marine, at any spatial resolution | Harfoot et al., 2014a |
| Integrated Assessment Models | Global and regional | Integrated | Multiple models | High | Interdisciplinary assessment | Harremoos and Turner, 2001; Tilman et al., 2001 |

#### 4.3.2.2 Predictability

No model can capture the full complexity of ecosystems and perfectly predict biodiversity patterns and ecosystem function as impacted by a suite of drivers, such as through climate change or habitat modification (Bellard et al., 2015). However, models are useful to synthesise data, evaluate alternative hypotheses, and provide projections about potential future states.

This is illustrated by the study of Bellard et al. (2012), who reviewed the approaches most commonly used for estimating future biodiversity at global and regional scales. They found that projections from the different approaches vary considerably, depending on method, taxonomic group, biodiversity loss metrics, spatial scales and time periods. Nevertheless, the overall projections from the majority of the models indicated that future trends for biodiversity were alarming. This reiterates a general finding from the IPCC, which is that projections from individual models should not be taken at face value. Instead, an ensemble approach accommodating uncertainty in multi-model prediction is required for interpreting trends and for comparisons between models. Comparisons that involve applying numerous models to evaluate a given policy question (e.g. related to the efficiency of alternative measures for minimising the impact of climate change) provide a means not just for evaluating uncertainty, but just as importantly for studying why the models produce different answers. This may indeed lead to feedback that impacts not just the individual models, but also the underlying theory that is used to develop the models (see Figure 4.7).

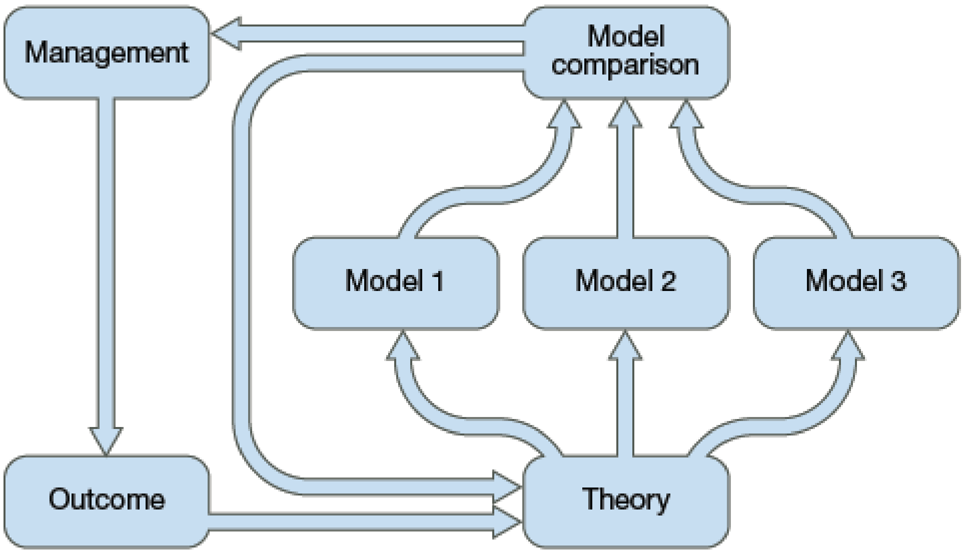


**Figure 4.7**: An overview of relationships between ecological theory, models, comparison and management. There may be numerous models to represent a given theory, and both the model comparisons and the management outcome may provide feedback to theory (Modified from Cuddington et al., 2013. *Process-based models are required to manage ecological systems in a changing world*. Copyright © 2013 by John Wiley Sons, Inc. Reprinted by permission of John Wiley & Sons, Inc).

It is becoming standard practice in many research fields for model fitting and statistical procedures to test model predictions on a known, typically simulated, data set in order to assess model behaviour and characteristics (e.g. in fisheries assessment, Hilborn and Walters, 1992). For models of complex natural systems, it is often not possible to test model predictions against simulated data, but a minimum requirement is that the models are 'validated' by a demonstration of each model's capability to at least exhibit the same behaviour as that which has been observed historically (Rykiel, 1996).

Validation here means consistency with observation (for instance as tested through time series fitting with formal information criteria evaluation).

As an example of a comprehensive model validation exercise, Elith and Graham (2009) constructed the distribution of an artificial plant species based on its affinity along three axes, related to preference for moisture, aspect ('southness') and geology, to obtain a 'true' spatial distribution for the plant. They constructed a spatial subsample of parameters (along the three axes), and used this to parameterize five different, commonly applied Species Distribution Models. By next predicting the full distribution for each method, they were able to validate model performance using true-false positive and negative patterns as well as the evaluation of predictions versus true values. This study, in addition to the direct evaluation of model performance, also demonstrated that model comparisons can be used to evaluate why different models give different predictions – which can be used for the further development of models as well as the refinement of ecological theory (see Figure 4.7).

While model comparisons are both needed and feasible, as demonstrated by the study of Elith and Graham (2009), they are difficult to conduct by any one research group as soon as the models involved are complex and in practice require both specific capacity and experience to be run optimally. For this reason, it is extremely important to build capabilities for inter-model comparisons, following in the footsteps of the Coupled Model Intercomparison Projects (CMIP) of the IPCC. Similar activities are now underway for biodiversity research as part of the Inter-Sectoral Impact Model Intercomparison Project (ISI-MIP), which is a community-driven modelling effort that brings together impact models across sectors and scales to create consistent and comprehensive projections of climate change impacts.

## 4.4 Modelling feedbacks and interactions

Both human and non-living environmental drivers influence biodiversity and ecosystem functions through a number of processes. In turn, biodiversity exerts feedbacks on both systems (Figure 4.8). Consideration of the feedbacks is important as they may cause non-linearity in interaction dynamics, which can potentially move a system beyond thresholds and tipping points (e.g. regime shift: Lenton, 2011).

Changes in biodiversity interact with different drivers of biodiversity change (e.g. climate change, disturbance regimes such as forest fires, invasive species and pests, and ecosystem processes) over different temporal and spatial scales. Changes in biodiversity and shifts in the distribution of plant traits can influence the climate at global and regional scales. For instance, General Circulation Models based on simulations indicate that the widespread replacement of deep-rooted tropical trees by shallow-rooted pasture grasses would reduce evapotranspiration and lead to a warmer, drier climate (Shukla et al., 1990). Similarly, the replacement of snow-covered tundra by a dark conifer canopy at high latitudes may increase energy absorption sufficiently to act as a powerful positive feedback to regional warming (Foley et al., 2000).

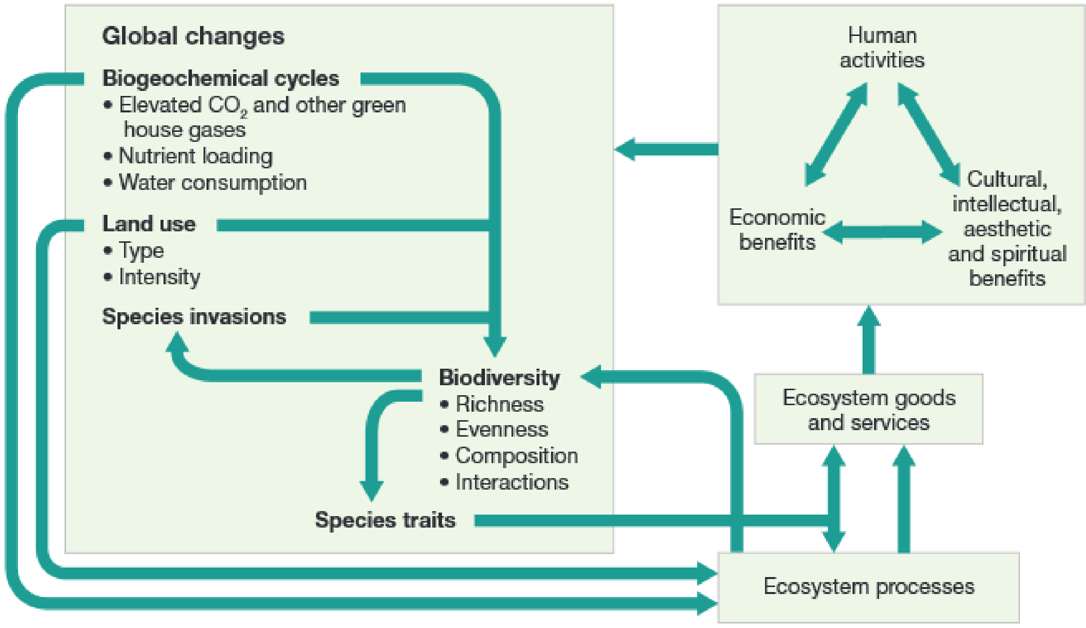


**Figure 4.8**: Schematic diagram of interactions between biodiversity, the human system and the non-living environment used for evaluating feedbacks related to species invasions. The figure represents feedbacks between biodiversity, drivers of biodiversity change and the interactions between these drivers (Modified by permission from Macmillan Publishers Ltd: [Nature] Chapin et al., 2000, 405, 234-242, copyright 2000).

Feedbacks between drivers and biodiversity or ecosystem levels usually involve a high level of complexity in the models because changes in state variables at different levels (either biological or others) should be able to interact and cause emergent dynamics. Changes in biodiversity, for instance, can impact disturbance regimes such as fire, which in turn are strongly determined by climate (Pausas and Keeley, 2009) and fire-suppression efforts (Brotons et al., 2013).

Biodiversity and ecosystem models as discussed in Section 4.3 describe the impact of abiotic drivers such as climate, nutrient cycling, atmospheric concentration of greenhouse gases including $CO_2$, water resources, fire, and land use on the biotic systems, including their biodiversity and ecosystem services. Many of the modelling approaches are capable of simulating the feedback of the biotic system on abiotic and human drivers as well. For example, many of the process-based models simulate carbon sequestration in vegetation and soils, and thus the impact on atmospheric greenhouse gas concentrations. Process-based models can also simulate feedbacks, from vegetation change to forest fires (LANDIS). Furthermore, many of the Dynamic Global Vegetation Models (ex-IBIS Foley et al., 1996; Kucharik et al., 2000; Sitch et al., 2003) are able to simulate feedback between the biotic system and water resources. However, only a few Dynamic Global Vegetation Models include detailed feedback to nutrient cycling. Dynamic Global Vegetation Models have been also used to study feedback between vegetation and past climate. General Circulation Models/ Atmosphere-Ocean General Circulation Models too include vegetation feedbacks to climate. Neither the process-based models (including Dynamic Global Vegetation Models) nor the General Circulation Models/ Atmosphere-Ocean General Circulation Models include the feedback of biodiversity and ecosystems to human societies. However, IAMs are capable of simulating impacts of changes in biodiversity and ecosystems on human systems, including economic activities.

## 4.5 Model complexity

Matching model complexity to policy and decision-making needs while keeping the model as simple as possible is a major challenge in the future development of biodiversity and ecosystem models (Merow et al., 2014). We here describe three general strategies that should help limit model complexity: model what matters, adopt hierarchical modular modelling approaches, and standardise protocols for model communication.

The first general strategy is the formulation of critical biological processes directly relevant to the question addressed or the problem to be dealt with. Avoiding unnecessary increases in model complexity requires a careful assessment of the biological processes that most directly affect species distributions at the spatial and temporal scales of interest for each particular study (Guisan and Thuiller, 2005). Although there is no general recipe to select the relevant biological processes, those related to species auto-ecology will always have a central role. Habitat selection and population dynamics in species-level models may be formulated with more or less detail, but are fundamentally important to predict species distribution dynamics (Willis et al., 2009; Kunstler et al., 2011).

Biological processes should only be modelled explicitly and internally (i.e. using process-based models) if they are critical for the question at hand. The remaining processes can be modelled externally and formulated into the model by means of input spatial layers or parameters modified by additional modelling frameworks (Smith et al., 2001). Such an approach may facilitate the flexible structuring of models by allowing sub-models to be plugged into one another (e.g. McRae et al., 2008). In this modular structure, the upper levels provide external contextual information (and hence external dynamics) to the lower ones. Hierarchical modular structures have the advantage of 1) being easier to integrate across different spatial and temporal scales (e.g. to downscale the results of processes formulated at higher levels (del Barrio et al., 2006)), and 2) being able to assess the levels of uncertainty added at each stage (Larson et al., 2004; Chisholm and Wintle, 2007). However, modularity may be limited for those target species that modify their environment or interact with other biotic entities (Midgley et al., 2010). Research is needed to compare the outputs of models with different degrees of complexity in the light of validation data appropriate to the process or driver under study (Roura Pascual et al., 2010). Only in this case will it be possible to build a body of reference regarding the minimum acceptable levels of complexity to analyse a given problem.

## 4.6 Accounting for uncertainty

Policymaking related to biodiversity and ecosystem functioning must take place based on the currently available knowledge. It must also be done recognising that uncertainty is associated with all science, including modelling, due to data limitations, the representation of processes, and the resolution of the ecosystem scale. Environmental complexity is an emergent property of the environment – it is not just that our models have limitations.

The fact remains that the environment is incredibly complex and interconnected. However, policymakers have to make decisions even in the face of uncertainty, to act on drivers in order to conserve ecosystems and biodiversity. To support decision making, models aim to synthesise this complexity into a reasonable number of dimensions.

In biodiversity and ecosystem modelling, the uncertainty arises from two primary sources: model uncertainty and uncertainty in the input parameters (or scenario uncertainty). Different models represent different physical processes differently, and to varying extents and levels of detail. This leads to model uncertainty. Input parameters, for example climate projections, add to the modelling uncertainty. An example of model uncertainty is that models generally do not take into account tipping points and non-linearity (Whiteman et al., 2013). Additionally, many models generally leave out the natural processes and feedbacks that are difficult to model given the current state of knowledge, even though these processes may cause large impacts. An example of uncertainty arising from input parameters is the uncertainty inherent in climate or land-use change projections. In addition, existing impact assessment studies – including the biophysical and integrated assessment models (IAM) – generally tend to work with the mean of the probability distribution of projected impacts, neglecting the low-probability, high-impact tails of the distribution (Weitzman, 2009; Ackerman et al., 2010; Marten et al., 2012). Impact studies generally focus on single-sector or single region-based assessments. The potential interactions among sectors and regions, which can adversely impact biodiversity and ecosystems, are therefore not adequately included in the quantitative estimates (Warren, 2011).

Similarly, the ambient policy and management practices and socio-economic stresses leading to the degradation of natural resources are also not included in most sectoral impact assessment models. Also, although key human-related issues such as armed conflict, migration and loss of cultural heritage have a lot of potential to impact natural ecosystems, impact assessment models do not include these human system-related stresses (Hope, 2013). IAM-based economic analyses of impacts are generally conservative, as these studies make optimistic assumptions about the scale and effectiveness of adaptation (Marten et al., 2012; Hope, 2013). In this section, we present the sources of uncertainty in models of biodiversity and ecosystems, some options to address uncertainty, and approaches to communicating uncertainty.

### 4.6.1 Sources of uncertainty

Link et al. (2012) and Leung et al. (2012) highlighted six major sources of uncertainty confronting ecosystem modellers (Figure 4.9).

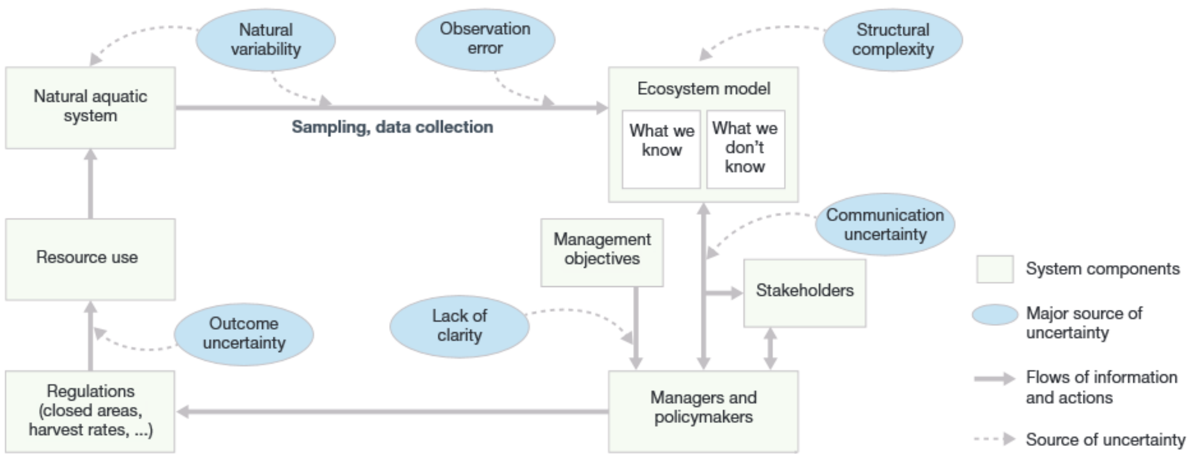


**Figure 4.9:** A conceptual diagram of the flow of information and actions in a typical Living Marine Resources management system. Rectangles represent components of the system, solid arrows indicate flows of information and actions between components, and ellipses represent major sources of uncertainty (Modified from Link et al., 2012. *Dealing with uncertainty in ecosystem models: The paradox of use for living marine resource management*. Copyright © 2012 by John Wiley Sons, Inc. Reprinted by permission of John Wiley & Sons, Inc).

#### 4.6.1.1 Natural variability

Natural variability or stochasticity includes biological differences among individuals, either within the same environment (genetic differences) or between environments (plasticity), differences among populations within a community, changes in spatial distributions with time, density-dependent or independent variation in a vital rate, seasonal or inter-annual variability in realised environmental conditions, or shifts in productivity regimes. Natural variability increases ecosystem model uncertainty by reducing the precision of parameter estimates.

#### 4.6.1.2 Observation error

Observation error is inevitable when studying organisms in either a single species or an ecosystem context (e.g. Morris and Doak, 2002; Ives et al., 2003, as cited in: Link et al. (2012)). The environmental characteristics of a particular area (even those that we can measure fairly accurately) are difficult to relate directly to the full experience of mobile organisms that move into and out of that area. Thus, natural variability can actually exacerbate observation error. Observation error adds uncertainty to ecosystem models through reduced precision, misspecified parameter distributions, and biased parameter estimates.

#### 4.6.1.3 Structural complexity

The structural complexity of a model arises from many factors, such as the number of parameters it includes; the number of ecosystem components and processes it simulates; the temporal scale; the nonlinearities, log effects, thresholds and cumulative effects incorporated in those processes; and whether or not it includes features such as spatial dynamics or stochasticity (Fulton et al., 2003). Structurally complex ecosystem models are gaining in use, in part due to improved computing capabilities and also due to the intricate, multi-sector, cross-disciplinary questions commonly being addressed in ecosystem-based management.

Ecosystem models are diverse in terms of scope and approach, but share the general feature of a large number of parameters with complex interactions. These models are necessarily built with imperfect information. Given these inevitable uncertainties, large and complex ecosystem models must be evaluated through sensitivity analyses with independent data before their output can be effectively applied to conservation problems (McElhany et al., 2010). Uncertainty in climate change scenarios arises from different greenhouse gas emission storylines and from differences between climate models, even if driven with the same storylines (McElhany et al., 2010). This can be partly addressed by using climate change scenario data from several emission storylines, but also by using results from multi-model studies (i.e. an ensemble of climate models). Process-based models are widely used to assess the impacts of climate change on forest ecosystems (McElhany et al., 2010). Climate change impact studies that do not integrate parameter uncertainty may overestimate or underestimate climate change impacts on forest ecosystems.

### 4.6.2 Options for reducing uncertainty

All model types carry multiple uncertainties, but there are potential options for reducing uncertainty, as discussed by Beale and Lennon (2011). It is important to establish the full range of model behaviours by carrying out a sensitivity analysis and considering different combinations of models and parameters. Sensitivity analysis is useful to determine the importance of each source of uncertainty. Apart from the sensitivity analysis of the model parameters, it is also important to consider the interaction between models and the data. Furthermore, running each model multiple times can assess the full range of model behaviour, parameter uncertainty and natural variability. One way of assessing uncertainty is to apply a mixed approach to uncertainty assessment comprising both the model and scenario uncertainty (Dunford et al., 2014). In addition, bifurcation points and decision nodes in models and scenarios need to be identified, and this should be supplemented by monitoring the system as it approaches these nodes to verify system behaviour. Monitoring can reduce the model and scenario uncertainty by adjusting the model in the light of the observations through a process of ‘data assimilation’.

One way of reducing uncertainty is to use multi-model ensembles (averages/weighted average), where it is suggested to avoid averaging model results unless the distribution of results suggested by all models is unimodal. Multi-model ensemble is not the only way of combining multiple model types, as different model types can also be joined statistically. For example, niche-based models and demographic- or process-based models could be integrated across spatial scales in a hierarchical framework or, more simply, Dynamic Global Vegetation Model output could feed into species distribution models to better predict the reliance of species on particular biomes.

### 4.6.3 Communicating uncertainty

An important consideration is the effective communication of these uncertainties when presenting assessment and modelling results. The purpose of the study strongly determines what uncertainty information is relevant and when to communicate uncertainty to policymakers and decision makers, and it is important to convey at least the robust main messages from a modelling assessment (Kloprogge et al., 2007).

The main challenge in developing a generic guideline for communicating uncertainty is that each assessment or decision-support context is unique. For example, in the case of species distribution modelling, Gould et al. (2014) report that the spatial distribution of uncertainty is not homogeneous and can vary substantially across the predicted habitat of a species, and that this depends on how the uncertainty impacts the model specification. Furthermore, modellers often encounter situations in which a number of potential sources of uncertainties cannot be quantified. In these situations, Gould et al. (2014) recommend that all potential sources of uncertainty should at least be systematically reported, along with model outputs.

Communicating uncertainty not only involves reporting on the uncertain aspects of the models themselves, but also provides insight into these aspects by elaborating on questions such as: Where do the uncertainties originate? What significance or implications do they have in a given policy or decision context? How might a reduction in uncertainty affect the decisions to be made? Can uncertainty be reduced? And how is uncertainty dealt with in the assessment or decision-support activity?

Communicating uncertainty to policymakers is different from communicating with scientists as far as the content and the form of presentation is concerned. Knowing the target audience and what matters to them is therefore important. Furthermore, the policy relevance of information on specific types of uncertainty depends on the phase of the policy cycle. Early in the cycle, for example, the focus would probably be on the nature and causes of a problem, while later on the focus may shift to the effects and costs of intervention options (Kloprogge et al., 2007).

It is important to adopt a systematic approach to the provision of information, for example through the ‘progressive disclosure of information‘ (PDI; Kloprogge et al., 2007). Under this approach, a report and associated publications are subdivided into several 'layers'. The 'outer' layer consists of the press releases, executive summaries, and so on. Here, it is advisable that non-technical information be presented with uncertainties integrated into the main messages and with the context emphasised. An example is the emphasis on the significance and consequences of assessment findings by the IPCC in summaries for policymakers. The 'inner' layers, comprising of appendices, background reports, and so on, can then provide detailed technical information and elaborate on the types, sources and extent of uncertainty. With regard to any of these layers, bear in mind when writing the purpose of the layer the purpose of the uncertainty communicated within it, the information needs of the target group, and the target group's expected interest in the layer. It is desirable that the target community's views are canvassed while designing the scenarios and recommendation as to what level of uncertainty is acceptable, both to the target community and scientifically.

## 4.7 Ways forward in biodiversity and ecosystem modelling

Modelling allows policymakers to assess the implications of scenarios of drivers and policy options for the future of biodiversity and ecosystems (Pereira et al., 2010). A diverse range of modelling approaches, from local to global scales, and from individual to ecosystem levels, have been developed to assess the impacts of direct drivers on biodiversity and ecosystem functioning and to investigate the feedback effects of biodiversity on these drivers. However, important challenges still remain in the link between biodiversity modelling and policymaking due to model complexity, uncertainty, and the lack of available data and knowledge (Mouquet et al., 2015).

Despite the availability of modelling approaches and applications developed in recent years, the biodiversity community needs to develop a common road map to better integrate predictive modelling with the challenges and needs derived from the current biodiversity crisis. A good example is seen in climate change research, where Global Circulation Models and Earth System Models have helped significantly in advancing understanding of the role of greenhouse gas emissions in driving the future climate.

Petchey et al. (2015) have introduced a road map for ecological predictability research. The road map describes the feedbacks and interactions between fundamental research on which the models are based, the data feeding into such models, and using evaluation of model outputs to inform development of new models, thereby improving the accuracy and usefulness of predictions. These feedbacks and interactions point to the need for an integrated approach to making models that meet the predictive requirements of stakeholders and policy (Figure 4.10).

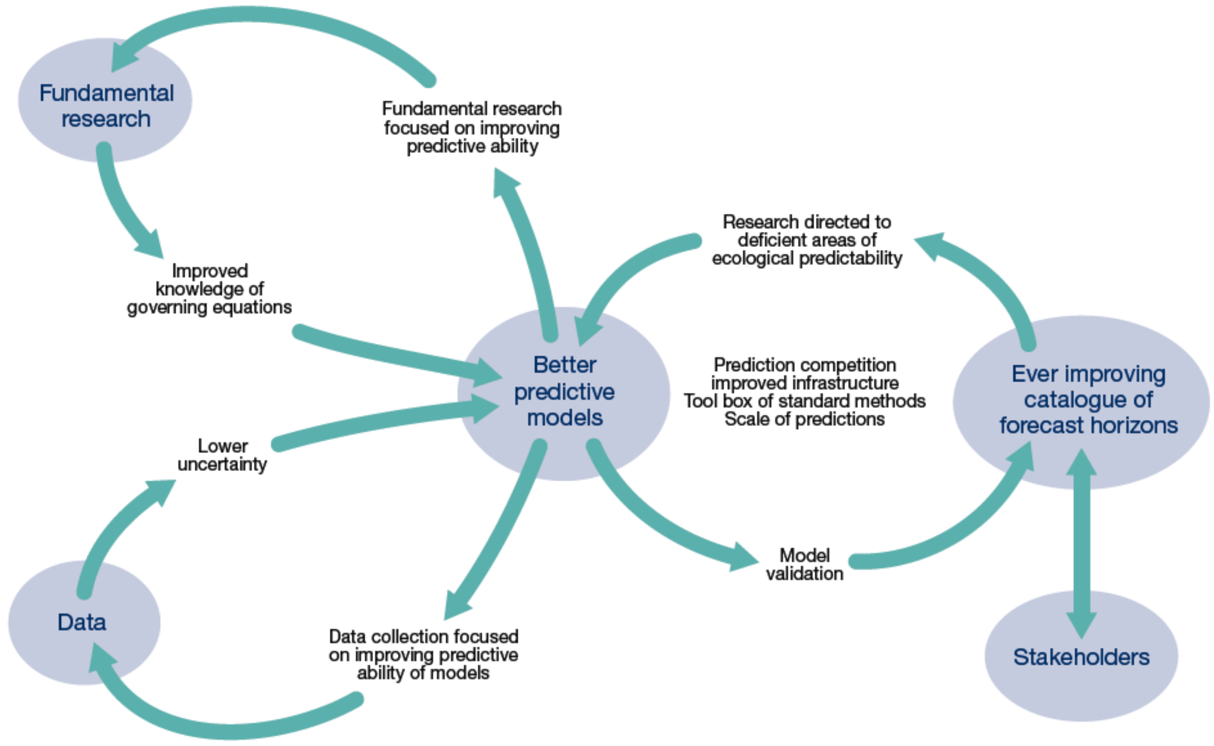


**Figure 4.10:** Schematic outline for improving model predictability in ecological research. The indirect interactions and feedbacks (e.g. between fundamental research and data and predictive models) are left implicit, yet are extremely important (Modified from Petchey et al., 2015. *The ecological forecast horizon, and examples of its uses and determinants.* Copyright © 2015 by John Wiley Sons, Inc. Reprinted by permission of John Wiley & Sons, Inc).

IPBES needs to recognise the complexities linking drivers of environmental change to biodiversity and ecosystem dynamics, and acknowledge the value of modelling as a method of producing a formal abstraction of such complexity and as a scientific tool for supporting decision making. When adequately framed, modelling approaches can be used as robust policy support (Guisan et al., 2013). However, IPBES also needs to keep in mind the significant capacity constraints and important gaps in the formalisation of the links between ecosystem models and policymaking. Therefore, future efforts should strongly encourage stakeholder participation as early as possible. This should be done to maximise the correspondence between the assessment objectives and the outputs and limitations of the modelling approaches (Guillera Arroita et al., 2015). Furthermore, the contextual interpretation of the modelling results and model uncertainty needs to be a joint activity of modellers and decision makers.

Finally, biodiversity and ecosystem modelling urgently requires adequate guidance regarding the typology of models used in isolation or combined in each of the assessments. Model intercomparison programmes should lead to increased collaboration among modelling groups and also with field ecologists to develop suitable protocols for modelling impacts of drivers on biodiversity and ecosystem functions, for example regarding scale, time frame, data collection and validation protocols, agreed processes, uncertainty analysis, and standardised outputs of the modelling studies. The promotion of model intercomparison groups will be vital for developing consistent protocols and standardised data, parameters and scenarios, as well as for incorporating long-term observation data and addressing and communicating uncertainty.

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

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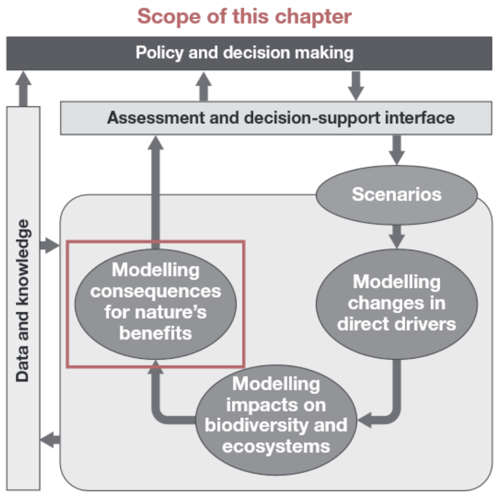
Chapter 5

## Table of contents

**Purpose of this chapter:** Describes the current state of ecosystem service models and modelling approaches for IPBES assessments and other users of ecosystem service models. Highlights the strengths and weaknesses of different approaches to modelling ecosystem services and critically reviews major types of ecosystem service models for generating outputs of relevance to different policy and decision-making contexts (as covered in Chapter 2).

**Target audience:** Aimed mostly at a more technical audience such as scientists and practitioners wanting to identify appropriate approaches to modelling ecosystem services for particular applications.




# Key findings

**The main contribution of an ecosystem service approach to decision making comes from considering bundles and trade-offs among multiple ecosystem services (5.2, 5.5).** Assessments of an ecosystem service in isolation can be useful for specific contexts, but assessing ecosystem services individually risks hiding trade-offs and synergies between ecosystem services that are often crucial in many decision-making contexts.

**Ecosystem service models are undergoing rapid development (5.4).** The number, diversity and application of ecosystem service models has greatly increased over the past decade. A variety of ecosystem service models exist, although most are limited in their ability to represent dynamic processes or social-ecological feedbacks. Consequently, the ability of ecosystem service models to project or analyse alternatives is weak, and most current models do not adequately represent the Intergovernmental Platform on Biodiversity and Ecosystem Services (IPBES) Conceptual Framework. There is also a lack of models that bridge multiple knowledge systems or connect to indigenous or local knowledge. Many new types of ecosystem service models are in development, and research 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 still in the preliminary stages, as is modelling the impact of changes in institutions and anthropogenic assets on ecosystem services (5.2, 5.5, 5.7).** Developing such tools will require investment and the transdisciplinary collaboration of policymakers 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 many ways in which human well-being can be assessed, and the study of human well-being is also rapidly developing.

**Modelling methods, tools and participatory processes each have particular strengths and weaknesses that make them appropriate in different decision contexts (5.4, 5.5).** Ecosystem service models can be useful as part of a process of developing, choosing, implementing and evaluating alternative ecosystem service strategies. 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 as they are easier to understand, use and assess. Simplicity is especially important in assessing multiple ecosystem services where reliable models of multiple services have not been well developed.

**Applying multiple models to the same case produces more robust decisions because different types of analyses and models are needed at different stages of the policy cycle (5.5).** Most real-world applications of ecosystem service models combine multiple modelling approaches and tools because no one modelling approach or tool is able to do everything well. Different phases in the policy cycle have different types of decision contexts and require different types of modelling tools. Furthermore, even within a phase of the policy cycle, multiple models are needed to analyse how changes in biological diversity and anthropogenic assets alter the benefits different people receive from nature.

**Models of biodiversity and models of ecosystem services are not well connected. Ecologists increasingly understand how biodiversity produces ecological functions (Chapter 4); however, many ecosystem services models utilise aspects of land use and land cover to predict ecosystem services (5.4, 5.5.3, Chapter 4).** 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 models, as would improving the understanding of the role of social and abiotic factors in mediating ecosystem services. Ecosystem services are produced by social and ecological factors in addition to biodiversity, so including all these aspects would likely increase the predictive quality of ecosystem service models. Which approaches yield the biggest improvements in model quality will likely depend upon the social-ecological context, data availability and the ecosystem services being considered.

## Key recommendations

**IPBES can help foster the development of a community of practice for ecosystem service modelling, data and scenario building and use (5.6, Chapter 7).** Ecosystem service research is currently fragmented and models and scenarios would benefit from more integration. Some modelling groups have developed a community of practice, but wider communities of practice need to be developed to support the use of models and scenarios in multiple regional assessments and to develop new models, scenarios and methods to better bridge multiple knowledge systems. IPBES could use its Task Force on Knowledge, Information and Data (Deliverable 1d) to facilitate access to scenarios, models and data by encouraging governments and scientists to make their models and data freely available using open access or creative commons licensing. Already available is the Ecosystem Services Partnership (ESP) spatial data mapping and sharing tool jointly developed by the European Commission Joint Research Centre and CSIRO (Commonwealth Scientific and Industrial Research Organisation) (see http://esp-mapping.net/Home/). This ESP tool allows users to upload and download spatial data on mapped ecosystem services and query the database on the data available for different ecosystem services and locations. IPBES could also use its Task Force on Capacity Building (Deliverables 1a/b) to promote, maintain and enhance communities of practice.

**IPBES can play an important role in promoting new ways to include multiple values and indigenous and local knowledge systems in models and scenarios (5.2, 5.5, 5.7).** 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, investment will be required in including multiple values and knowledge systems in models and scenarios. It is important that IPBES ensures that the Task Forces on Capacity Building (Deliverables 1a/1b), Indigenous and Local Knowledge (Deliverable 1c) and Knowledge, Information and Data (Deliverable 1d) and the Expert Group on Values (Deliverable 3d) facilitate communication among these communities as well as the development of new model and scenario approaches.

**Thematic, global and regional assessments of ecosystem services (IPBES Deliverables 2b, 2c, 3b) would benefit from the analysis of outputs from models of ecosystem services at multiple scales (5.3.1, 5.5).** 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 the 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. Global and regional assessments should consider linking and analysing connections among multiple cross-scale ecosystem service assessments that include models of local ecosystem service dynamics.

**Regional assessments of ecosystem services (IPBES Deliverable 2b) could link and analyse connections among multiple cross-scale ecosystem services based on models of local ecosystem service dynamics (5.4.3).** Local models of ecosystem services are better developed than regional or large-scale models of ecosystem services. Therefore, regional assessments should strongly consider integrating and comparing multiple local models of ecosystem services as opposed to relying primarily on regional-scale 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 and identifies the substantial gaps within this research. 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. The chapter concludes with an assessment of gaps and recommendations for actions and future research that would develop the capacity to make better use of ecosystem services and human well-being models in IPBES.

## 5.2 The IPBES Conceptual Framework and knowledge for modelling ecosystem services and human well-being linkages

This chapter focuses on how people have used models and scenarios to understand how ecosystems contribute to human well-being. The contribution of ecosystems to human well-being is strongly shaped by social institutions and anthropogenic assets. Most of Earth's ecosystems have been reshaped, restructured and reorganised by people. This shaping has occurred intentionally, through ecological engineering (such as terraced rice paddy agriculture to enhance the availability of desired benefits), as well as unintentionally from unintended by-products of other actions (such as the impact of climate change on ecosystems). Social activities or conditions as well as biophysical dynamics directly impact ecosystems and human well-being (Butler and Oluoch-Kosura, 2006; Fremier et al., 2013). The IPBES Conceptual Framework integrates these pathways of interaction among people and nature (Figure 5.1). This chapter builds on Chapters 2 to 6 in this assessment that address related parts of the IPBES Conceptual Framework. Chapter 2 focuses on the decision contexts in which models of biodiversity and ecosystem services are used, Chapter 3 on drivers of changes in nature, Chapter 4 on modelling impacts of these drivers on biodiversity and ecosystems, and Chapter 6 on how to integrate multiple models. The following section explains how models and scenarios of nature's benefits to people relate to the IPBES Conceptual Framework.

### 5.2.1 Ecosystem services, human well-being and the IPBES Framework

The development and implementation of policies and practices that ensure and enhance the flow of ecosystem services to people require the inclusion of ecosystem services in decision making. There are a wide variety of contexts in which decisions are made concerning ecosystem services, and effectively including ecosystem services in these decisions requires different types of models and scenarios that can identify how social and ecological change alter the dynamics of ecosystem services and human well-being. In the context of IPBES, models and scenarios are essential components of IPBES regional and thematic assessments, and can enable dialogue and communication among the broader IPBES community. Furthermore, the IPBES Conceptual Framework recognises that different knowledge systems will conceive of nature's benefits in different ways that go beyond the use of different ecosystem services but include different conceptualisations of access, decision making, knowledge generation and knowledge itself (Díaz et al., 2015; Houde, 2007). This plurality is a challenge to modelling approaches that utilise a fixed model, but can be well incorporated by modelling and scenario approaches that enable dialogue among different people through participatory processes (Davies et al., 2015).

Nature provides multiple benefits to human societies. Throughout this chapter, these benefits are referred to as ecosystem services (MA, 2005b). These benefits are not produced in isolation, because the ecological and social processes that produce ecosystem services interact with one another so that actions to increase the supply of one ecosystem service often impact other ecosystem services (Bennett et al., 2009).

Furthermore, ecosystem services are produced at different scales, which means that no single scale is well suited to manage all ecosystem services, making it difficult to avoid conflicts and interactions between ecosystem service providers and beneficiaries. The benefits from ecosystem services are diverse and unevenly studied and conceptualised. Best understood are the flows of food and materials people receive from nature. Much more poorly understood are the ways in which nature stabilises and regulates environmental variation; how nature contributes to people's cultural, psychological and spiritual well-being; and how biodiversity provides options and opportunities for people in the future.

Demand for ecosystem services is increasing even as the intensified human modification of Earth's ecosystems is reducing the capacity of these ecosystems to continuously provide these benefits (Cardinale et al., 2012). Paradoxically, despite the simplification, conversion and degradation of many ecosystems, the past century has seen consistent and global increases in health, life-expectancy, education and income (UNDP, 2014). Much of the global simplification of ecosystems has been to replace diverse ecosystems with those producing high levels of agricultural ecosystem services, and to date this conversion has been considered to enhance rather than decrease human well-being (Raudsepp-Hearne et al., 2010b). In addition, even simplified ecosystems can provide diverse ecosystem services, for example research has shown that while the conversion of diverse tropical forest into monoculture oil palm plantations has eradicated a multitude of biodiversity and reduced ecosystem services, it has not eliminated many preferred services and has enhanced others (Abram et al., 2014). Furthermore, during the past century social innovation and technical advancement have increased the productivity of ecosystem services, allowing more benefits to be obtained with less impact.

Scientists have identified many aspects of how ecosystem services contribute to human well-being, but how biodiversity, anthropogenic assets, institutions and culture shape these links is only starting to be understood.

Nature's benefits to people are strongly shaped by the dynamic interaction of anthropogenic assets, biodiversity and ecosystems, and institutions (Figure 5.1). Nature's benefits are typically unequally distributed among different sectors of society or beneficiaries. The relationships between people and ecosystem services have been conceptualised in a variety of ways prior to the IPBES Conceptual Framework (Figure 5.2), and different conceptual frameworks have highlighted either a linear flow of nature's benefits to people, or a cyclical interaction between people and nature. While the IPBES Conceptual Framework is cyclical, the practice of ecosystem service assessment and modelling has often been linear. Circular conceptualisations better capture the reality of interactive feedbacks between people and nature, but linear conceptualisations are much easier to analyse and operationalise.

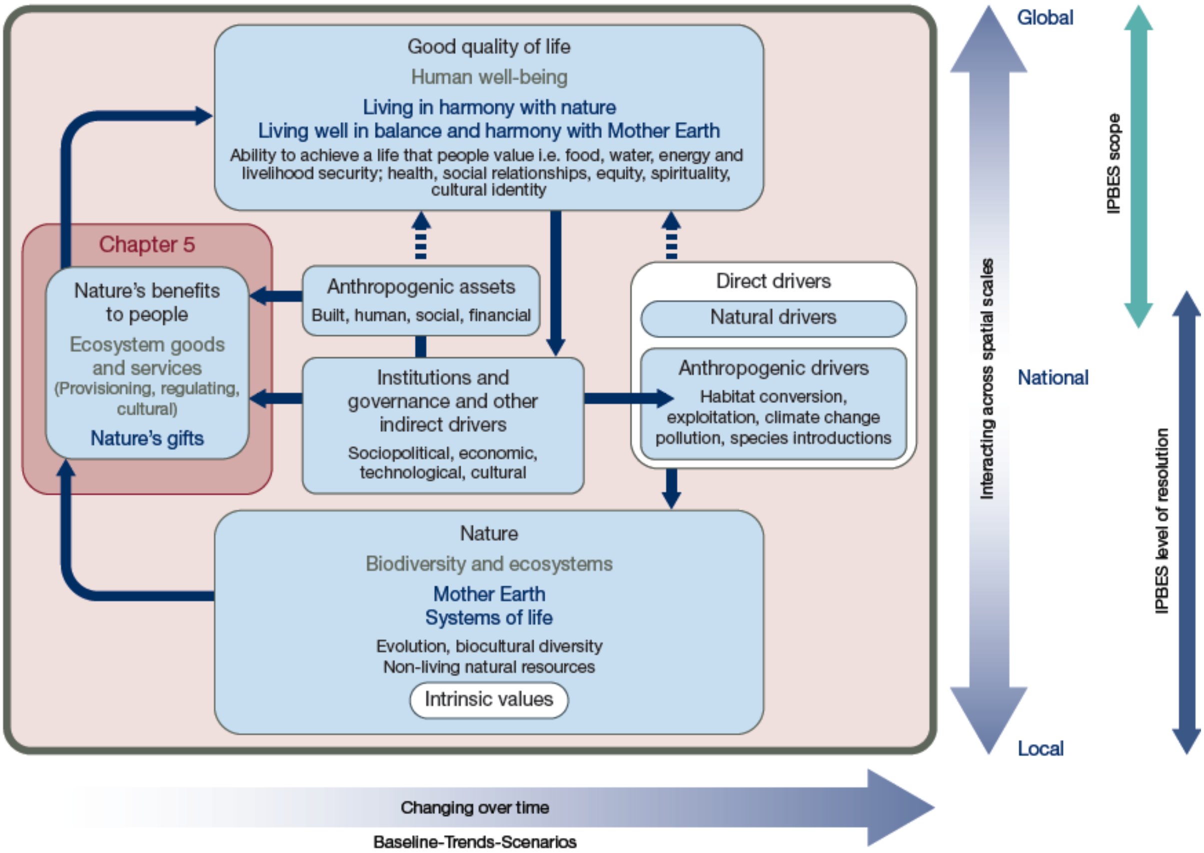



**Figure 5.1:** This chapter focuses on approaches to modelling how nature provides benefits to people, and how these benefits are influenced by nature, institutions, anthropogenic assets, and natural and anthropogenic direct drivers (Modified from Díaz et al., 2015).

Ecosystem service modelling approaches need to be able to represent the key features of these conceptual frameworks in ways that are useful across a variety of decision contexts. The IPBES Framework and others emphasise the importance of the following three considerations in ecosystem services and human well-being modelling: 1) scales (e.g. local, national, regional and global, and scale transferability), 2) interactions (e.g. how institutions shape access to ecosystem services), and 3) feedbacks (e.g. mutual reinforcing interactions between ecosystem services and human well-being). Additionally, landscape spatial patterns and temporal dynamics usually need to be considered to model ecosystem services in any real-world processes. While models need to be able to represent the temporal and spatial complexity of ecosystem services, it is also important they can be used as tools for learning and bridging between different knowledge systems. These goals of learning and bridging are both important to IPBES and are essential to the practical use of ecosystem service models in decision contexts involving diverse stakeholders or contested issues.

However, no existing ecosystem services and human well-being modelling tools or approaches capture all these dynamics (scales, interactions and feedbacks), and combining different models and tools to better incorporate these dynamics remains an area that requires further research and development (Carpenter et al., 2009).

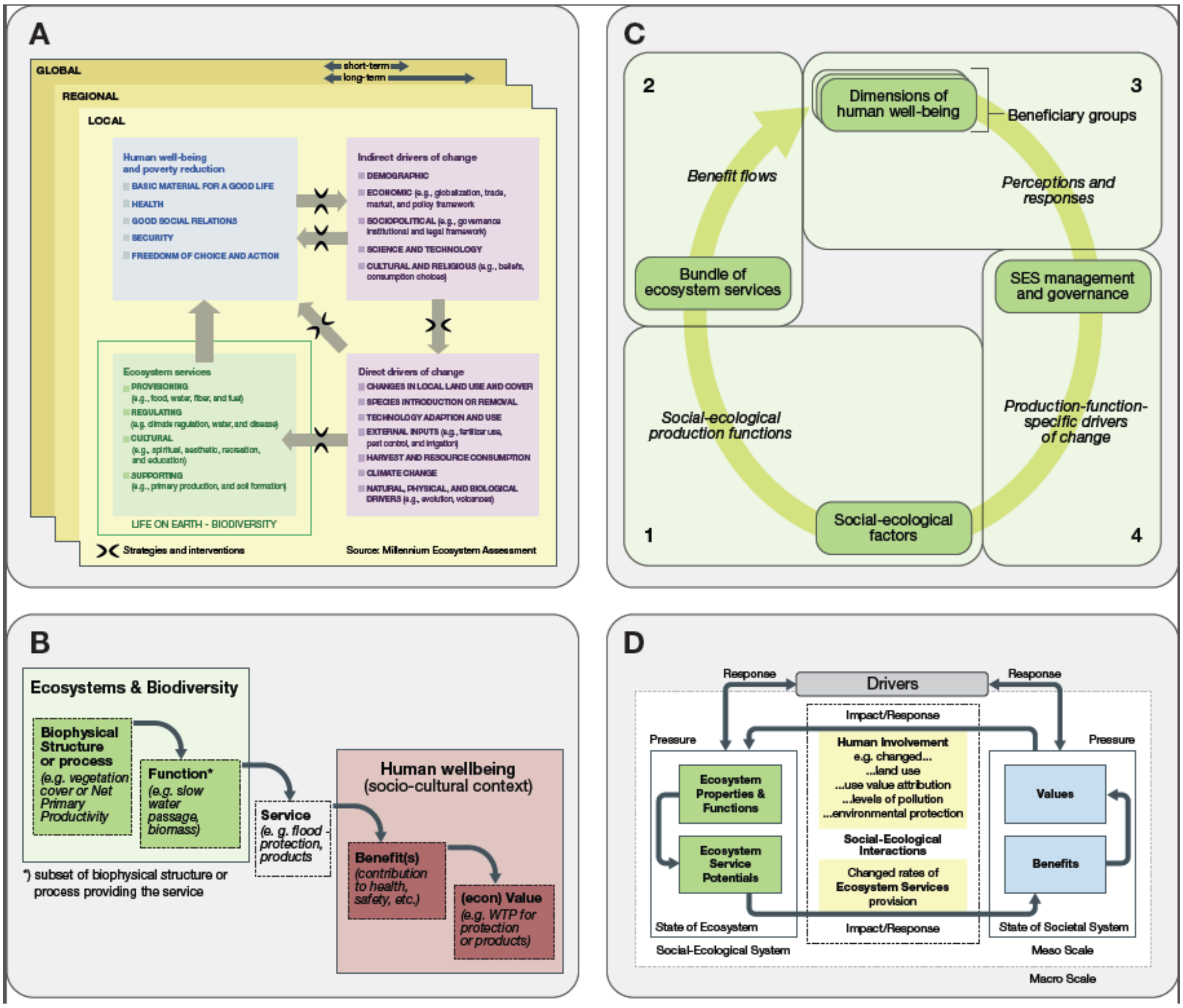



**Figure 5.2:** There are many ways in which natural and anthropogenic assets have been conceptualised as producing ecosystem services. Many of these conceptualisations have a flow of benefits from nature to society, while others emphasise the co-creation of benefits by nature and society. (A) The Millennium Ecosystem Assessment Conceptual Framework (Modified from MA, 2005b. All rights reserved World Resources Institute); (B) the cascade model of ecosystem services (De Groot et al., 2010); (C) an SES approach to identifying social-ecological factors and interactions highlights the importance of dynamic feedback between society and ecosystems (Modified from Reyers et al. (2013). *Getting the measure of ecosystem services: a social–ecological approach.* Copyright © 2013 by John Wiley Sons, Inc. Reprinted by permission of John Wiley & Sons, Inc); and (D) an integration of the ecosystem service cascade (B) and social-ecological perspective (C) (Modified from Nassl and Löffler, 2015, DOI: 10.1007/s13280-015-0651-y, http://creativecommons.org/licenses/by/4.0/).

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

Scenarios and models can improve decision making by transparently representing assumptions underpinning decisions, compressing and synthesising complex information in an understandable way, identifying unexpected outcomes, and exploring alternative policies.

The value and utility of a model has to be judged against the context in which it is being used. We describe these situations as decision contexts (Chapter 2).

The attributes of the decision context will determine the scope of ecosystem service modelling and scenario analysis required. For example, the approaches that are useful for resolving disputed forest governance, decision-making processes, management and logging between indigenous peoples and national government will be different from approaches that are useful for helping a city plan highway development in an urban wetland, or for helping government marine planners attempt to craft policies that benefit conservation as well as sport and commercial fisheries.

A decision context can be defined by numerous variables describing biophysical, social, economic, spatial and temporal dimensions (Chapter 2, Table 5.1). Important attributes of decision contexts relevant to ecosystem service modelling include: temporal and spatial extent and resolution, ecological complexity, political (jurisdiction and administrative) scale, socio-cultural characteristics of stakeholders (knowledge systems, value pluralities), governance and institutional settings, and the decision objectives and scope. A decision context determines what types of modelling approaches are appropriate for a given situation. For example, modelling approaches that are transparent and enable participation, exploration, interaction and negotiation may be more applicable when there is disagreement and divided governance. The best types of models in these situations will be the simpler expert opinion and correlative models which are typically participatory. Alternatively, more technical and data-intensive approaches such as process-based and integrative models may be more appropriate for decision contexts that have clear objectives, a sound system understanding and non-contested governance (Table 5.1).

**Table 5.1:** Decision contexts (from Chapter 2) and how they relate to modelling strategies. For each aspect of a decision context, different types of modelling approach are needed for easier-to-decide or more-difficult-to-decide modelling contexts. The modelling strategies columns show focuses or approaches recommended for easier or more difficult decisions relevant for each row.

| | Decision-context attributes | Easier-to-decide | More-difficult-to-decide | Modelling strategies for easier decisions | Modelling strategies for more difficult decisions |
|---|---|---|---|---|---|
| **Governance** | *Actors* | Single/executive | Multiple/negotiated | Optimisation | Multi-criteria |
| | *History* | History of governance | Novel governance | Predictive | Explorative |
| | *Legitimacy* | Accepted | Contested | Implementation | Conflict resolution |
| | *Sectors* | Single | Multiple | Single model | Integrated models |
| | *Participation* | Consultation | Decision | Visualisation | Group interaction |
| **Decision** | *Decision time horizon* | Short term (months) | Longer term (decades) | Extrapolate | Feedbacks |
| | *Decision frequency* | One-off | Repeated | Adaptive | Robust |
| | *Objectives* | Single | Multiple | Securing benefits | Managing trade-offs |
| **Stakeholders** | *Values* | Homogenous | Diverse | Assumed | Negotiated |
| | *Knowledge system* | Homogenous | Diverse | Single | Multiple models |
| **Ecology** | *Heterogeneity* | Single ecosystem | Multiple ecosystems | Single model | Multiple models |
| | *Diversity* | Single species | Multi species | Populations | Food webs |
| | *Flows across landscape* | Weak connections | Strong connections | Independent | Feedbacks |
| | *Stochasticity* | Low and predictable | High and unpredictable | Deterministic | Stochastic |
| **Scale** | *Cross-scale dynamics* | Weak external influence | Strong external influence | Focus system | Driver/system |
| | *Temporal extent* | Short-term | Long-term | Correlative | Processes |
| | *Temporal grain* | Seconds | Millennia | Data | Drivers |
| | *Spatial extent* | Local | Global | Data availability | Data accuracy |
| | *Spatial grain* | Metres/seconds | Kilometres/degrees | Detailed data | Aggregated data |
| **Information** | *Scientific knowledge* | High | Low | Predictive models | Exploratory models |
| | *Data availability* | High | Low | Precise models | General models |
| | *Scientific capacity* | High | Low | New models | Model adapted |

The policy cycle (Chapter 2) can be used to define four stylised types of decision-making context: agenda setting, policy design, implementation and review (see Chapter 2). Agenda setting involves identifying and defining the features of a problem. Policy design involves the formulation of rules and regulations to guide actions to address that problem. Policy implementation is a process of organising, prioritising and scheduling activities to achieve articulated goals. Review involves navigating the inevitable tensions, trade-offs and opportunities that emerge from implementing plans and policies. Scenarios and models can improve decision making by 1) transparently representing key processes and assumptions that underpin decisions; 2) compressing and synthesising complex information in an understandable way; 3) helping identify unexpected outcomes; and 4) testing and exploring new policies and assumptions.

Decision contexts are also shaped by social, ecological and biophysical factors. Social factors include 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 by other people impacted by the decision. The ecological factors include what aspects of nature are considered in a decision (e.g. water, a population of a species, or an interacting ecosystem), what ecosystem services are considered, and how ecological variables are being considered in the landscape or seascape.

Many decisions involving biodiversity, ecosystem services and human well-being are complex and morally fraught. Decisions may rely on poorly understood ecological processes and involve conflicting interests and values among different groups in society who may have different worldviews, wealth and power. Structured decision making can help improve such processes, but there is likely to be disagreement about which decision process is legitimate as well as which decisions should be made (see Chapter 2). For example, in the management of a coastal fishery, commercial fishers, indigenous groups, environmental groups and government bodies are likely to disagree over who makes decisions, how they are made, and the boundaries of decision making (Peterson, 2000).

In this chapter, we relate different approaches to modelling ecosystem services to different phases in the policy cycle and different types of decision contexts. We also specifically address the needs of ongoing IPBES regional and sub-regional assessments. Below, we briefly outline the major aspects and aims of IPBES regional and sub-regional assessments as well as the likely decision contexts for the major aspects of assessments.

### 5.3.1 IPBES regional and sub-regional assessments

The IPBES regional and sub-regional assessments (IPBES/3/6, http://ipbes.net) assess five major aspects of biodiversity and ecosystem services, as follows:

1. Trajectories of nature's values: possible changes in 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;
2. Trajectories of ecosystems: the status and trends of biodiversity and ecosystem services, including the structural and functional diversity of ecosystems and genetic diversity;
3. Trajectories of drivers: the status and trends of indirect and direct drivers and the interrelations of such drivers;
4. Risks: future risks to drivers, biodiversity and ecosystems, ecosystem services and human well-being under plausible socio-economic futures;
5. 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 Convention on Biological Diversity (CBD).

The assessments are currently being undertaken for four regions (Africa, the Americas, Asia Pacific, Europe and Central Asia), with each regional assessment following a common structure but tailored to regional-specific contexts. The regional assessments aim to answer policy-relevant questions such as 1) the contribution of biodiversity and ecosystem services to economies, livelihoods and well-being; 2) the status and trends of these biodiversity and ecosystem services; 3) the pressures driving change in these biodiversity and ecosystem services; and 4) possible interventions to ensure the sustainability of biodiversity and ecosystem services (IPBES/3/6/Add.1, http://ipbes.net). The IPBES global assessment will build on the regional and sub-regional 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; however, these assessments will require many different approaches to modelling and scenario analyses. For example, the decision contexts for influencing trajectories of nature's economic values to humans are grounded in the social, geographical and economic sciences, and will be defined primarily by the importance of ecosystem service flows to beneficiaries. These analyses typically focus on geopolitical boundaries at scales relevant to people and are shaped by available demographic data. Decisions impacting substantially on beneficiaries will likely be made at a 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 (Bagstad et al., 2014; Reyers et al., 2013; Syrbe and Walz, 2012). Recent concepts for linking beneficiaries to ecosystem services include quantifying service provisioning and benefitting areas and service connecting regions. For example, Renaud et al. (2013) clearly demonstrate the value of ecosystems to people by showing how ecosystems can reduce risks associated with natural disasters. The questions asked within IPBES may include how to identify ecosystems of high scenic beauty and recreational value and the users of these areas (Palomo et al., 2013; Palomo et al., 2014). Also important is the location of communities most vulnerable to climate change, who could be beneficiaries of carbon sequestration as well as climate regulation services, and the location of communities most vulnerable to natural disasters such as flooding, landslides and cyclones, who could be beneficiaries of flood regulation, erosion control and extreme event moderation ecosystem services. 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).

While many social, cultural and political dynamics shape access to ecosystem services and exposure to hazards, much of the recent progress in ecosystem science focuses on developing biophysical models that aim to represent the processes that underlay the supply of ecosystem services, and the changes in supply from changes in ecosystems and biodiversity. Decisions will often be location-specific and 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. It is also important to identify new methods for incorporating social-ecological feedbacks and how social dynamics shape exposure to hazards and access to ecosystem services.

The questions asked in relation to the trajectories of ecosystems may include understanding the efficacy of land or water management interventions for improving the condition of ecosystems and 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 and vegetation community), although it may extend to landscapes if ecological connectivity is of interest, which will require the collective involvement of a highly diverse group consisting of many decision makers.

Another decision context of assessments aims to understand the drivers of 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. A scenario is a plausible and often simplified description of how the future may develop, based on a coherent and internally consistent set of assumptions about key driving forces and their relationships (MA, 2005a). The development of scenarios has often been used to bridge knowledge, information and data derived from multiple knowledge systems.

IPBES could both use existing scenario analyses and develop its own scenarios. There are many different types of scenario analysis that vary in terms of being exploratory or normative, expert-led or participatory, as well as in the scale at which they are conducted. 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) explored 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. Similar work has been carried out 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 (Bryan et al., 2014; Nelson et al., 2013). Other scenario analyses have adopted a narrative-based approach to include processes that are not modelled or well understood, such as shifts in diet or immigration policy, to be integrated with quantitative models of ecological or climate dynamics (Oteros-Rozas et al., 2015). These approaches have also been used to incorporate indigenous and local knowledge systems into scenarios. Additionally, the archetypes or families of global scenarios identified in Table 6.3 could be used to structure or focus IPBES global assessments.

Overall, IPBES can use scenarios to bridge knowledge systems, integrate disparate models and data, evaluate policy, and focus scientific investigation and synthesis.

## 5.4 Types of models

A variety of approaches and tools have been used to assess and model ecosystem services. This section reviews different approaches and tools for modelling ecosystem services. It starts by describing correlative (5.4.2), process-based (5.4.3) and expert-based (5.4.4) approaches to modelling ecosystem services, then compares these general approaches (5.4.5). The creation of scenarios is included as part of the section on expert-based modelling. More widely used ecosystem service modelling tools such as the Integrated Model to Assess the Global Environment (IMAGE), Integrated Valuation of Ecosystem Services and Trade-offs (InVEST) and Ecopath with Ecosim (EwE) are discussed and compared (5.4.6), and the section concludes with a brief discussion of economic approaches to ecosystem services (5.4.7). The following section (5.5) assesses how well these different approaches match different decision contexts.

### 5.4.1 Types of attributes that differentiate ecosystem services models

Modelling the impacts of change in biodiversity and ecosystems on beneficiaries takes many different forms for many different purposes (Crossman et al., 2013b).

Models can be classified as correlative, process-based and expert-based (Chapter 4).

Each of these different modelling approaches have strengths and weaknesses that make them a better fit to different types of decision contexts (Cuddington et al., 2013). In real-world problems, different types of models are often used in combination or are integrated (Chapter 6).

While ecosystem service models are available in all of these categories, most general models of ecosystem services tend to be either correlative models of ecosystem services that use land cover and land use to predict ecosystem services, or process-based models that simulate biophysical processes and typically arrive at production functions and a detailed system understanding (Crossman et al., 2013a; Kareiva et al., 2011; Maes et al., 2016). Many local or regional ecosystem service assessments use expert-based models (Davies et al., 2015), while ecosystem services have also been included in the global integrated assessment model IMAGE (Stehfest et al., 2014), which combines expert, correlative and process-based models. Most models of ecosystem services do not explicitly model changes in biodiversity (Chapter 4), but use correlative or expert-based approaches to estimate how changes in ecosystem services correspond to changes in land cover and land use that implicitly cause and result from changes in biodiversity.

Different models focus on different parts of the IPBES Conceptual Framework. Most often modelled is the supply side of ecosystem services, in other words the dynamics of the flow of services from nature to people. Much less common is modelling changes in beneficiaries' demands for ecosystem services: for example, how changes in human populations' income or preferences translate to changes in demand for the flow of services (for a discussion of drivers of biodiversity and ecosystem change see Chapter 3). While biophysical processes and some economic processes are often modelled by process-based models, changes in anthropogenic assets and institutions have been typically modelled using correlative or rule-based models.

The following section summarises the potential benefits of the correlative, process-based and expert-based ecosystem service modelling approaches (Table 5.2) and briefly describes the attributes, dynamics, scales, levels of complexity and handling of uncertainty typically found in these models.

**Table 5.2:** Potential benefits of alternative modelling approaches, ranging from low (*) to high (***). (Modified from Cuddington et al., (2013) *Process-based models are required to manage ecological systems in a changing world.* Copyright © 2013 by John Wiley Sons, Inc. Reprinted by permission of John Wiley & Sons, Inc).

| Potential benefit | Correlative | Process-based | Expert-based |
|---|---|---|---|
| Model availability | *** | *** | *** |
| Evaluate alternative management strategies | * | *** | * |
| Transparent assumptions | *** | *** | * |
| Integrates and distributes uncertainty | *** | *** | * |
| Appropriate for projection | * | ** | * |
| Allows rapid response | ** | * | *** |
| Lower data requirements | * | * | *** |
| Ease of development | ** | ** | *** |
| Can incorporate social and cultural factors | * | ** | *** |
| Ease of bridging knowledge systems | * | * | ** |

## 5.4.2 Correlative 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 ecosystem services. For example, spatial data on perennial vegetation extent have been used to estimate the flow of ecosystem services such as the 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 in correlative models for other regulating services such as erosion prevention (Maes et al., 2012b).

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 can be specified as 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 of change produced in an output from an additional unit of input, all else held constant. However, such production functions will often fail when change is substantial, or when they are used in contexts in which key social-ecological factors are different from those in which they have been parameterised.

Simpler correlative models have improved with the addition of complexity by spatially disaggregating land use/cover data and combining these data with additional information (e.g. expert knowledge and higher spatial or temporal resolution data). Although still correlative-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 are complemented by a number of additional datasets is the study by Schulp et al. (2014a), which modelled the production and consumption of wild foods in Europe. As a proxy for production, Schulp et al. (2014a) 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. (2014a) used a mix of internet and literature searches, ingredient lists from cookbooks and hunting statistics. A related approach seeks to identify types of social-ecological systems producing different types of bundles of ecosystem services (Raudsepp-Hearne et al., 2010a). This approach uses covariance among multiple ecosystem services across a landscape to identify characteristic patterns of ecosystem service production.

### 5.4.2.1 Strengths and weaknesses

The strength of correlative models is that they are simple and easy to apply, but the weakness is that – because they are not based on process understanding – they can dramatically fail in novel or data-poor situations.

The relative simplicity of correlative models means that they require fewer resources and less technical expertise. This simplicity makes correlative models useful where ecosystem service data and understanding are lacking, but their extrapolations should be treated as initial assessments. However, their simplicity does make them very amenable to participatory processes. Correlative models are transferable, as in the highly influential Costanza et al. (1997) study and recent follow-up (Costanza et al., 2014), which estimated the supply and value of the world's ecosystem services across a handful of broad global biomes. However, the credibility of correlative models has been questioned because of their generalisation across non-similar contexts (Eigenbrod et al., 2010). Typically absent in correlative methods are system dynamics such as socio-ecological feedbacks, complex interactions, temporal changes and the inclusion of external drivers of change. When these dynamics are important, or expected to play a strong role, correlative models may produce inaccurate results.

## 5.4.3 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 can estimate the flow of ecosystem services from natural capital with more realism than correlative models. Process models can include socio-ecological feedbacks and interactions at fine scales, and therefore are highly suitable for assessing 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 (Bryan et al., 2014; Paul et al., 2013). 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, and have often not been well integrated or reported in the ecosystem services literature. For example, hydrologists have for decades been modelling complex hydrological processes using detailed 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).

#### 5.4.3.1 System Dynamics Models

Integrated system dynamics models have also been used to translate biodiversity and ecosystem properties into ecosystem services and benefits, within the context of large-scale feedbacks between natural capital and human-made capital. One of the earliest of these models was the Global Unified Metamodel of the Biosphere (GUMBO) (Boumans et al., 2002), 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 et al., 2015). Documentation for these models is only partially available, and they were developed using Simile software, a commercial software package, for which the code is not publically available. This current lack of documentation and code makes the models difficult to adapt, reuse or verify; however, models developed in Simile can be exported for use in open source software and MIMES developers envision that the models will be available for online collaboration development in the future (Boumans et al., 2015).

A wide variety of other system dynamics models have and are being constructed to address particular types of ecosystem service trade-offs, or the complex dynamics of particular places. For example, the Land-Use Trade-Offs (LUTO) model is a complex model (Bryan et al., 2014) that integrates the results of other models of Australian land-use change to explore economic trade-offs among agricultural provisioning services, carbon sequestration and biodiversity conservation. The model is available, but not particularly accessible (e.g. there is no documentation available on the internet). Others have used systems approaches to model linkages between the economy, society and the environment, where flows of ecosystem services provide value to both the economy and society. For example, Fiksel et al. (2014) developed a triple value model to represent dynamic linkages and resource flows among these three systems as a way to characterise sustainability; however, this model was implemented in Vensim, which is a commercial modelling toolkit, and the model is not publically available. System dynamics models are a powerful way of representing complex systems with their feedback and interdependencies, and of examining trade-offs. The development and accessibility of these models in open source software is an important area for future research.

#### 5.4.3.2 Strengths and weaknesses

The strength of process-based models is that they represent a scientific understanding of key dynamics, which can enable learning and enrich decision assessment, while their chief weakness is that they require substantial knowledge and time to develop.

Process-based models are designed to mechanistically represent key system dynamics, which enables them to include key ecological and social feedback processes and to evaluate alternative future management scenarios in complex situations. They can be calibrated and validated with observed data and assessed through sensitivity and uncertainty analysis. However, they typically require substantial time to create and use. The complexity of system dynamics models and frequent lack of clear or publically available documentation often limits their use, modification and verification by others. Even when such models are freely available, the difficulty of understanding how the models function restricts their use to the modellers who initially created the model. Therefore, they are often not easily transferable to other locations, except by their creators. They require detailed technical expertise to create, and cannot easily be used, analysed or modified by non-experts.

### 5.4.4 Expert-based models

Social-ecological dynamics are often complex, integrated and poorly understood. Models of social-ecological dynamics often need to integrate disparate types of data and expert knowledge in the absence of mechanistic theory or quantitative data. A variety of 'soft systems' approaches have been used to model ecosystem services, including Bayesian belief networks, fuzzy cognitive maps, social-ecological scenarios and matrix models. Eliciting and using expert knowledge can be challenging, especially when multiple knowledge domains or systems are used, and all these types of models require that expert knowledge be used in a fair, rigorous and efficient way (Drescher et al., 2013).

#### 5.4.4.1 Bayesian belief networks

Bayesian probabilistic models can be used to integrate expert knowledge with multiple data sources to model the flow of ecosystem services (Haines-Young, 2011; Landuyt et al., 2013). Although not in themselves models that simulate biophysical processes, Bayesian models call or take outputs from biophysical models (correlative and/or process-based), which they then integrate with probabilistic qualitative data often derived from expert knowledge about social systems. Their ability to integrate expert and stakeholder knowledge with quantitative data and models makes Bayesian models very useful for comparing alternative scenarios (Keshtkar et al., 2013; Fletcher et al., 2014) in situations of limited data availability and/or where there are participatory and/or co-design requirements. Landuyt et al. (2013) provide a review of 47 uses of Bayesian belief networks to assess ecosystem services. Advantages of this approach include the ability to combine different types of data and include new data and its explicit treatment of stochastic uncertainty, both of which makes it useful for applications with limited data. However, it difficult to evaluating feedback processes, and the translating multiple types of data can be complex and confusing. Bayesian models have been proposed as a robust way to bridge the gap between the more accurate but less transferable and participatory process models, and the simple and transferable but heavily generalised correlative models (Landuyt et al., 2013).

#### 5.4.4.2 Fuzzy cognitive maps

Fuzzy cognitive maps are similar to Bayesian belief networks because they combine an identification of causal links with probabilistic estimations of their impact. These models aim to capture the interactions among variables in the absence of detailed data. These models are typically developed from discussions with experts, then iteratively revised into a model structure and function which corresponds to shared expert knowledge. For example, Daw et al. (2015) used a fuzzy cognitive map to link detailed simulation models and qualitative scenarios in a participatory workshop on coastal ecosystem services. These models can be used to make qualitative scenarios more rigorous and to elicit models from diverse groups of people (Kok, 2009). They have similar strengths and weaknesses to Bayesian belief networks.

#### 5.4.4.3 Social-ecological scenario analysis

Scenario analysis is a type of soft systems modelling that is increasingly used to analyse the dynamics of social-ecological systems, with a strong focus on ecosystem services and human well-being (Peterson et al., 2003, Oteros-Rozas et al., 2015). Scenario analysis differs from traditional quantitative models in that it is flexible and accessible, and can integrate non-quantitative, partially quantitative, or fully quantitative information (Amer et al., 2013). Social-ecological scenarios usually analyse how decisions or policies perform across alternative futures in a way that addresses uncertainties both by improving the social capacity to consider and shape the future and by identifying robust policies (Bennett et al., 2003; Carpenter et al., 2006a). As frameworks for integration, scenarios provide a platform for addressing and bridging different approaches to knowledge, to views of how the world works, and to 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 knowledge scenarios. Prior to, but particularly since the Millennium Ecosystem Assessment (MA), a diversity of participatory social-ecological scenarios has been run in many different places around the world (Oteros-Rozas et al., 2015). These projects range 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., 2008). 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 and can also be used for communication and outreach, thus enriching the understanding of social-ecological dynamics, uncertainties and options (Peterson et al., 2003).

Compared with technical models, scenarios are often more accessible, integrative and engaging; they are also better able to explicitly address trade-offs among different groups and multiple pathways between ecological change and human well-being (Carpenter et al., 2006b). However, scenarios are less rigorous, less comparable, and less generalisable than technical models. Non-participatory scenarios can often be created and analysed quickly, similar to simple expert-based models. However, participatory scenario processes take longer and require an effort similar to participatory modelling exercises (Oteros-Rozas et al., 2015). Many global assessments, as well as some smaller scenario exercises, have taken an iterative approach to quantitative models and qualitative storylines. In large assessments, this story and simulation approach (Alcamo et al., 2005) allows multiple complex integrated assessment models to be combined, but this requires substantial amounts of coordination and expertise. This 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 on how to conduct social-ecological scenario planning projects have been developed, but the accessibility, diversity and guidance on tools and techniques for scenario process management and scenario development require further improvement. Recent research has focused 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), and the better use of softer quantitative modelling approaches such as fuzzy cognitive maps (Jetter and 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 (Oteros-Rozas et al., 2015).

#### 5.4.4.4 Matrix models

Matrix models are a common way of integrating expert opinion, land-cover data and other empirical data. Combining maps of land cover and land cover's contribution to ecosystem services using Geographic Information Systems (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 (Burkhard et al., 2009; Jacobs et al., 2015). These models 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). They have gained popularity as a pragmatic way of quantifying spatio-temporal changes in the supply of multiple ecosystem services under scenarios and drivers of environmental change, especially in data-sparse locations (Kaiser et al., 2013), and of meeting the co-design, participatory and transdisciplinary needs inherent in ecosystem service assessments (Fish, 2011; Jacobs et al., 2015).

These features of matrix models are well illustrated by an example from Cape Town, South Africa. Responding to requests from urban politicians and land managers, O'Farrell et al. (2012), scored urban ecosystem services from remnant native vegetation in Cape Town, in which ecosystem service production by different land-use classes was estimated based on expert opinion. They then analysed historical and potential future land-cover change and diffuse spatial benefits away from remnant vegetation. This approach 1) allows the relatively cheap and rapid identification of key areas and issues, 2) enables useful discussions between ecosystem service experts and urban managers, and 3) facilitates an analysis of changes in land management or differences among particular sites (O' Farrell et al., 2012).

#### 5.4.4.5 Strengths and weaknesses

The significant strength of expert-based models is that they allow the relatively easy incorporation of diverse types of expert knowledge into ecosystem service models. This strength is particularly useful for work that seeks to bridge multiple knowledge systems. However, this strength comes with the weakness that expert knowledge is often partial, biased and can be incorrect, especially when applied to novel, complex or highly uncertain situations.

This weakness can be partially reduced by ensuring that the process of eliciting expert opinion is transparent, that it tracks where knowledge comes from, and that it includes multiple, diverse sources of expertise. Furthermore, expert-based models can easily be combined with process-based or correlative models to allow the strengths of those models to be realised while including knowledge that is not available otherwise. The challenge is to develop easier, more effective, more transparent methods to combine these different types of knowledge in integrated approaches such as participatory social-ecological scenarios.

### 5.4.5 Comparing modelling approaches

Approaches to modelling ecosystem services vary in their analytical strengths and weaknesses as well as in the time required to apply them (Table 5.3). Correlative models focus on existing statistical relationships, matrix models focus on spatial patterns, fuzzy cognitive maps and Bayesian belief networks focus on representing expert knowledge, dynamic systems models focus on feedbacks and interactions among people and nature, and participatory scenarios combine beliefs, feedbacks and interactions based on the integration of expert knowledge with models. Correlative and matrix models are also easy to use and relatively easy to create and revise, making these types of models useful initial or rapid assessment tools. System dynamics models are difficult to produce and less accessible but allow interactions and slow dynamics to be explored. Participatory scenarios can combine many types of knowledge relatively effectively and can be easily revised, but are difficult to verify or translate from place to place.

Some of these approaches are combined to complement one another in many local ecosystem assessments, but the development of more standardised, tested and evaluated methodologies for integrating multiple methods is required (Chapter 6).

**Table 5.3:** Comparing different types of modelling methods of ecosystem services. Many models mix different types of approaches. Participatory social-ecological scenarios often combine expert-based models with other types of models.

| Modelling approach | Temporal dynamics | Model type | Ease of use | Time to learn | Beneficiaries | References |
|---|---|---|---|---|---|---|
| Correlative models | No | Correlative | Easy | Medium | Multiple | Schulp *et al.*, 2014a |
| System dynamics models | Yes | Process | Hard | High | Multiple | Boumans *et al.*, 2015 |
| Bayesian belief networks | No | Expert | Easy | Medium | Multiple | Haines-Young, 2011 |
| Fuzzy cognitive models | No | Expert | Easy | Medium | Multiple | van Vliet *et al.*, 2011 |
| Matrix models | No | Expert | Easy | Low | Single | Burkhard *et al.*, 2009 |
| Social-ecological scenario analysis | Yes | Expert + | Easy | Medium | Multiple | Oteros-Rozas *et al.*, 2015 |

## 5.4.6 Description of major ecosystem services modelling tools

Examples of some of the major models and modelling approaches for quantifying ecosystem services are compared in Table 5.4; widely used modelling tools are described more fully below. Ecosystem service models are rapidly developing, therefore this cannot be a comprehensive assessment of all available models. We have placed more emphasis on models and modelling frameworks that are open access and well documented with a substantial community of practice.

### 5.4.6.1 InVEST: Integrated Valuation of Ecosystem Services and Trade-offs

InVEST is a well-developed and widely applied suite of models for different types of ecosystem services, typically using the spatial extent and configuration of habitat or land use as predictors of ecosystem services production. InVEST has been continually developed and expanded by the Natural Capital Project since 2006 (Kareiva et al., 2011). As of late 2014, the toolkit includes 16 distinct InVEST models suited to terrestrial, freshwater and marine ecosystems. InVEST models typically rely on simplified representations of biophysical processes 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 either in biophysical terms – whether absolute quantities or relative magnitudes (e.g. tons of sediment retained or percentage change in sediment retention) – or in 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 and focus on scenario inputs 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 for 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 and an active online community forum. 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, 2015). 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.

Feedback is not explicitly built into the model structure but is 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. (2015) concluded that these simple production function models are 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 the design of payments for ecosystem services (PES) schemes. The application of InVEST for ecosystem services assessment is most effective when it is embedded in a broadly-participatory iterative science-policy process (Rosenthal et al., 2014).

InVEST models run as stand-alone software tools, but users will need GIS 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, for example it will typically take between one and three people two months to a year to compile data and run one or more InVEST models, although this depends on the project scope and data availability. 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 models 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, the spatial extent of recreational kayaking, water quality, the number of recreational homes and habitat quality – under baseline conditions and scenarios of industry expansion and conservation zoning. They found that 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, with 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 found that stakeholders considered using different currencies for valuing different ecosystem services to be an acceptable approach.

#### 5.4.6.2 ARIES: ARtificial Intelligence for Ecosystem Services

ARtificial Intelligence for Ecosystem Services (ARIES) is a modelling platform incorporating multiscale 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 inclusion of 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 the source and use of a service. ARIES explicitly includes the flow of services to groups of beneficiaries using agent-based models (Villa et al., 2014), which is useful for considering trade-offs and for guiding policy (Bagstad et al., 2014).

ARIES is complex, and significant time and skill are required for independent applications of ARIES, which are likely to require the involvement of the ARIES development team as new users must be registered to use the platform. Bagstad et al. (2013a) compared an ARIES and an InVEST simulation for the San Pedro river, and estimated that the applications took 800 and 275 hours respectively. The model has significant data requirements; however, the ARIES system assists users in locating appropriate datasets. The ARIES team envisions developing generalised global models available in future releases, which will make ARIES more accessible. However, substantial changes in the structure and use of ARIES have reduced its ability to develop a community of practice. ARIES does not include valuation, although 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 and can be used with other ecosystem system service models, but requires the use of GIS.

#### 5.4.6.3 Ecopath with Ecosim (EwE)

Ecopath with Ecosim (EwE) was developed to dynamically represent energy flows through marine and aquatic ecosystems. Its structure means that it can easily include fishers and fish consumers in its models. It is one of the few ecosystem service models that explicitly represents both species and specific groups of beneficiaries; however, it can only assess limited sets of – usually fisheries-related – ecosystem services.

EwE consists of three interlinked components: Ecopath, Ecosim and Ecospace (Christensen and Walters, 2004). 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. Ecosim allows the time-dynamic simulation of ecosystems that are described by Ecopath and is based on an Ecopath model to provide some of the initial-state Ecosim parameters. It uses a system of differential equations 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). The spatial resource use of predators and preys is implicitly represented. It is primarily designed to explore fishing scenarios and their implications for the exploited ecosystems and fisheries catches. Ecosim also models the impact of environmental forcings, such as climate change and non-trophic interactions between functional groups. EcoSpace allows the 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.

EwE has been widely used to generate scenarios of changes in or the 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 limitations to beach seine fisheries on the well-being of coastal communities in Mombasa, Kenya, with a particularly focus on the poor (Daw et al., 2015). Specifically, EwE provided expected ecological and fisheries responses of the Mombasa coral reef and seagrass ecosystem under a range of fishing effort scenarios. The model represented trophic interactions between 56 functional groups of fish and the effects of 5 different types of fishers using different gears, including beach seine, fish trap, spear, hook and line, and net. Simulations provided indicators of food production, profitability and conservation as well as catch per unit effort by gear and fish type. The outputs from the EwE models were used to explore human well-being implications, using a 'toy' fuzzy cognitive map model (5.4.3.2) to combine the key linkages of fish abundances and catches with the well-being of individual stakeholders. The 'toy' model was used in a participatory workshop in which groups of stakeholders in the region were asked to explore ways to manage the fishing effort of different gear groups that would maximise the well-being of specific fishing gear groups or seafood traders (Daw et al., 2015). There are other modelling approaches for estuarine/marine fisheries that represent additional complexity, for example the ATLANTIS model (Fulton et al., 2014). ATLANTIS uses a similar framework, but is time- and data-intensive to apply and is used by a smaller community of practice.

#### 5.4.6.4 IMAGE 3.0

IMAGE 3.0 is an integrated assessment modelling framework developed to analyse the dynamics of 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 food provision from agricultural systems, water availability, carbon sequestration and flood protection. Estimation of the ecosystem services of wild food provision, erosion risk reduction, pollination, pest control, and attractiveness for nature-based tourism requires additional environmental variables and relationships (Maes et al., 2012a; 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 with estimates of the minimum quantity required by people to assess surpluses and deficiencies. This translates, 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.

#### 5.4.6.5 Other ecosystem service toolkits

Corporate Ecosystem Services Review (ESR), the Toolkit for Ecosystem Service Site-based Assessment (TESSA), Co$ting Nature and the Land Utilisation and Capability Indicator (LUCI) are other tools that are used to quantify ecosystem services and that are based on expert or correlative models that relate ecosystem state to ecosystem services but do not include valuation.

ESR (Hanson et al., 2012), developed by the World Resources Institute, is a structured methodology that helps businesses that interact with ecosystems to identify business risks and opportunities. ESR uses a qualitative approach to consider the 27 ecosystem services given in the MA. TESSA is a toolkit that uses decision trees to guide users through a process to rapidly prioritise ecosystem services for assessment and identify data needs and communication approaches. It provides a template that users must adapt to specific cases (Birch et al., 2014; Peh et al., 2014). Co$ting Nature (Mulligan et al., 2010) is an easily accessible web-based tool that can be used to estimate the costs of maintaining four ecosystem services (carbon storage, water yield, nature-based tourism and natural hazard mitigation) under scenarios of climate or land-use change. It includes detailed spatial data of the entire world, spatial models of social and biophysical processes, and scenarios of climate and land-use change. Quick assessments can be made with included data, or deeper analysis using locally produced data. LUCI is a tool similar to Co$ting Nature. As of late 2015, it is being revised into a second generation tool (lucitools.org). LUCI uses simple algorithms and outputs to identify and communicate ecosystem service trade-offs to stakeholders and decision makers. It is focused on agricultural landscapes and ecosystem services such as production, carbon, flooding, erosion, sediment delivery, water quality and habitat, based on GIS land and soil information (Jackson et al., 2013).

These toolkits are useful for providing an assessment of ecosystem services, either as a starting point for deeper modelling or scenario work or as a mechanism to connect to other ongoing analyses. Three of these approaches (ESR, Co$ting Nature and LUCI) are compared in Bagstad et al. (2013b), which assessed that ESR and Co$ting Nature were well-suited for immediate application for scoping or assessment, and that LUCI had significant potential but was not at that time adequately documented or supported for widespread use.

#### 5.4.6.6 Strengths and weaknesses

The widely used ecosystem service tools vary substantially in their focus, approach and user community. Due to the differences in scale, approach and ecosystem service focus, all the models have the potential to complement one another.

In particular, the easy-to-use rapid assessment tools can be used to provide preliminary assessments to guide deeper modelling or scenario work.

Most of these tools are only weakly dynamic and do not incorporate ecological or social feedbacks. IMAGE and EwE both include feedbacks, but IMAGE's ecosystem service models are quite simple and EwE's focus is narrowly on fisheries-related ecosystem services. While these tools are all useful for assessing current ecosystem services, and the impact of marginal changes on those services, they are not well suited for addressing transformative change or long-term trends.

**Table 5.4:** Summary of major ecosystem services model tools. Dynamic models are in orange, while snapshot models are in blue.

| Tool | Model type | Scale in space, time | Ease of use | Community of practice | Flexibility | Reference |
|---|---|---|---|---|---|---|
| IMAGE | Process | Global, dynamic | Difficult | Small | Low | Stehfest et al., 2014 |
| EcoPath with EcoSim | Process | Region, dynamic | Medium | Large | High | Christensen et al., 2005 |
| ARIES | Expert | Region, dynamic | Difficult | Small | High | Villa et al., 2014 |
| InVEST | Process and correlative | Region, static | Medium | Large | Medium | Sharp et al., 2014 |
| Co$ting nature | Correlative | Region, static | Easy-medium | Small | Medium | www.policysupport.org/costingnature |
| TESSA | Expert | Region, static | Easy | Small | Low | Peh et al., 2014 |
| Corporate ecosystem services review | Expert | Region, static | Easy | Small | Low | Hanson et al., 2012 |
| LUCI | Correlative | Region, static | Easy | Small | Medium | www.lucitools.org |

### 5.4.7 Green accounting

There are a number of accounting frameworks that make explicit the contribution of ecosystems and their services to economic activity. The expansion of economic accounting to include the environment is typically referred to as green accounting (Smulders, 2008), but has other names such as triple bottom line accounting and green GDP (Boyd and Banzhaf, 2007). Green accounting approaches need to align with economic accounting approaches to avoid the double-counting of ecosystem services in national economic accounts. Green accounting approaches have moved from including natural capital in national accounts to the explicit consideration of multiple ecosystem services.

Creating green national accounts has been approached in a number of different ways. Economic metrics that are ecologically adjusted by natural capital depletion can be produced by adjusting aggregated monetary measures of economic performance in response to impacts of changes to ecosystems, such as the adjusted Net National Product (Barbier, 2012). Inclusive or comprehensive wealth accounting estimates changes in natural capital along with measures of produced human and social capital. The key idea behind wealth accounting is that the future consumption possibilities, which include non-market benefits such as some ecosystem services, depend on the various capital types or asset base of a nation (Arrow et al., 2012; Dasgupta, 2009; UNU-IHDP and UNEP, 2014; World Bank, 2011). Other measures of economic welfare that incorporate changes in the environment that impact on human well-being include 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, including environmental components, in human well-being into a single value to represent advances in human development, such as the Organisation for Economic Co-operation and Development's (OECD) Better Life Index (OECD, 2015).

Other approaches to green accounting have measured the economic impact of alterations to ecosystems in space and time. The United Nations (UN) System of Environmental-Economic Accounting (SEEA) and the SEEA Experimental Ecosystem Accounting (SEEA-EEA) extend conventional economic accounts to the environment (EC, 2014; UN et al., 2014). The SEEA-EEA assesses the contribution of nature to economic and other human activities by organising biophysical data, estimating ecosystem services and tracking changes in ecosystem assets to nations (UN et al., 2014). The United States Environmental Protection Agency's (US EPA) new National Ecosystem Services Classification System (NESCS, U.S. EPA, 2015) is a good example of how these approaches can represent ecosystem services (Figure 5.3). It is an approach designed to standardise the classification of ecosystem services to simplify their valuation, include anthropogenic assets, and clearly link ecosystem services to specific beneficiaries. It focuses on the final ecosystem services (e.g. Boyd and Banzhaf, 2007), which avoids double-counting services – necessary for accurate economic valuation. NCSES is based on principles of accounting systems for economic goods and services, such as the North American Industry Classification System.

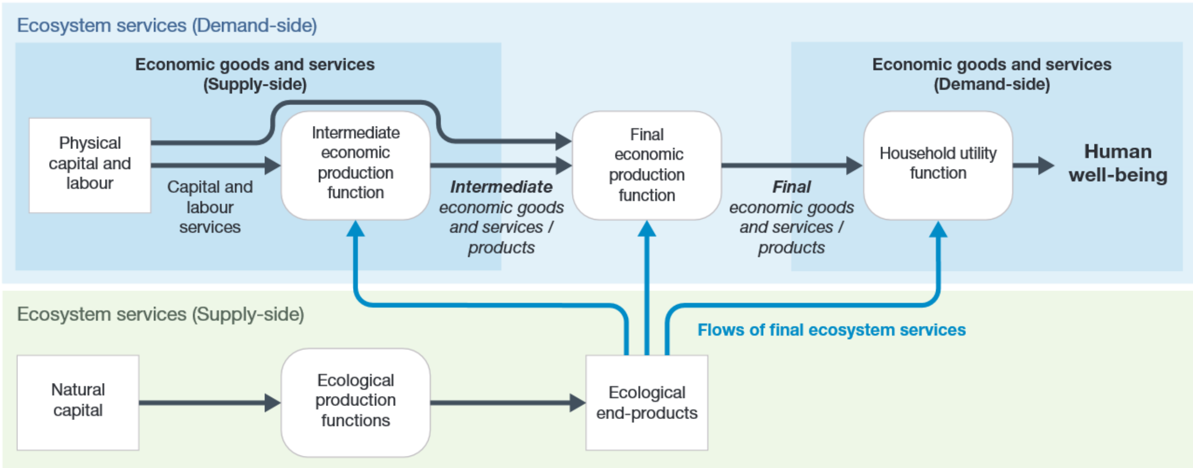



**Figure 5.3:** Conceptual framework for the US EPA National Ecosystem Services Classification System that includes the flows of final ecosystem services as inputs into human systems (Modified from U.S. EPA, 2015).

### 5.4.7.1 Strengths and weaknesses

Green national accounts can produce an aggregate picture of how a nation is doing, but they do not disaggregate benefits and costs among different beneficiaries. Other accounting methods can link changes to individuals, but like other ecosystem service models these approaches are better at capturing immediate or marginal changes rather than a longer period of systemic transformations as they do not account for multiple social and ecological feedbacks. These accounting approaches also do not currently capture more complex aspects of human well-being, such as the influence of nature on health and well-being. Green accounting approaches are rapidly developing as researchers attempt to better define practical definitions and measures of human well-being and incorporate the value of nature in national and other accounts, and they have the clear potential to complement and be combined with other ecosystem service modelling approaches.

## 5.5 Comparing model types across decision contexts

The variety of approaches and tools for modelling ecosystem services can be compared against how they perform in different types of decision contexts (Table 5.1). Most models of ecosystem services are focused on making decisions, but models can also be used as tools for dialogue, learning or evaluation. One way of simplifying the variety of decision contexts in which models can be used is the policy cycle (see Chapter 2). The policy cycle can be conceptualised as consisting of four related phases: agenda setting, policy design, implementation and review. While real-world decision making usually does not follow this idealised sequence of stages, the policy cycle helps organise discussion of different types of models. As what decisions are being made, about what, and by who is clear in some phases of this cycle and unclear and contested in others, different approaches to modelling are required.

Modelling needs vary across the policy cycle (Figure 5.4), but most existing work on ecosystem services is focused on the policy design stage of the cycle, while the implementation, review and agenda-setting stages are underemphasised.

Most ecosystem service models and modelling tools are designed to evaluate alternatives, but often assume a clear system definition, a lack of social-ecological feedbacks, and a unified, uncontested decision-making process. While existing tools have been used as part of agenda setting, they have generally not focused on enabling people to bridge different knowledge systems. As Martinez-Harms et al. (2015) note, the assessment of trade-offs and the prioritising of management actions are key steps in decision making for ecosystem services. However, learning and dialogue are also key aspects of decision making that have been relatively neglected by ecosystem service models. 'Soft systems' modelling methodologies, in particular social-ecological scenarios, have been used as learning tools. It is perhaps not surprising that ecosystem service models have not been more widely developed for learning or implementation as ecosystem services are relatively new in their application. However, there is a lot of potential to develop new modelling approaches to accelerate and improve the design of tactical models, and to develop open libraries of models and data, as well as methods for evaluating and comparing ecosystem service models and approaches to enable learning.

The role of models in the ecosystem service assessment process is sometimes conceptualised as a process from assessment to valuation (e.g. Bagstad et al., 2013b). However, this approach solely focuses on the elaboration of the design part of the policy process, and ignores the key role of the other parts of the process that determine what is designed, who gets to decide what is designed, and how a design is actually implemented in the real world.

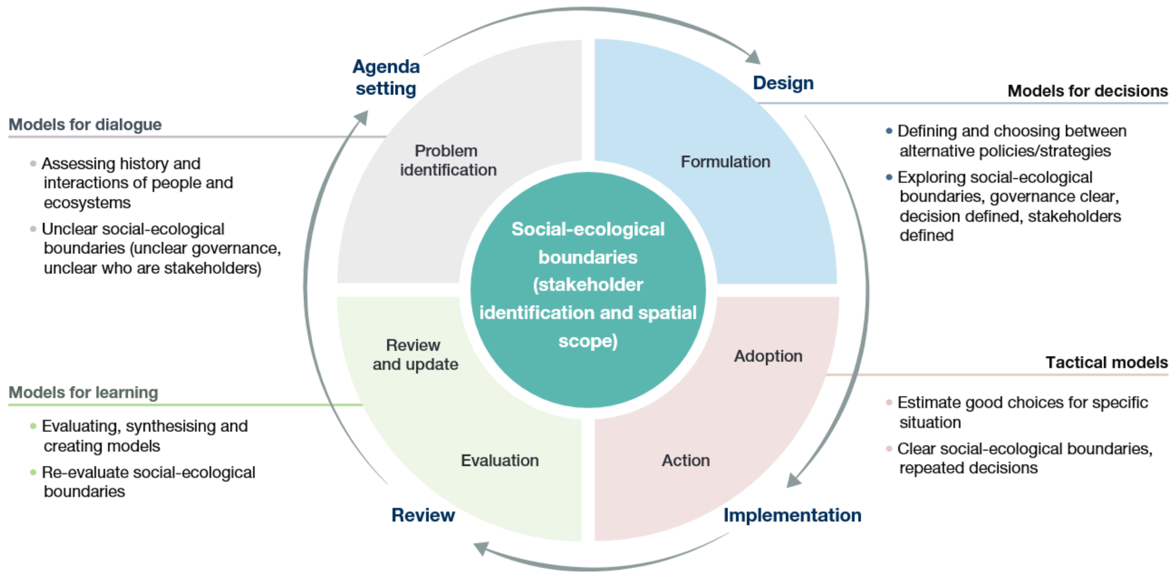



**Figure 5.4:** How different types of models (in bold) relate to a representation of the policy cycle (Figure 2.1). The policy cycle is conceptualised as agenda setting, policy design, implementation and review. While real-world decision making usually does not follow this idealised sequence of stages, the policy cycle helps organise discussion of different types of models. Because what decisions are being made, about what, and by who is clear in some phases of this cycle and unclear and contested in others, different approaches to modelling are required.

Participatory modelling approaches are particularly important in the agenda-setting and learning parts of the policy cycle.

These parts of the policy cycle typically feature decision contexts in which a problem is unclear or contested, so that modelling often needs to be done in a participatory fashion. A variety of these approaches were assessed to make different useful contributions to ecosystem service assessments (Davies et al., 2015). In particular, Davies et al. (2015) found that system dynamics modelling and Bayesian belief networks, if used in isolation, have a low likelihood of generating the trust and mutual understanding necessary for people to bridge their different knowledge systems. However, more participatory modelling, including group model building, can enable people to create more integrated, shared models of ecosystem services (Figure 5.5).

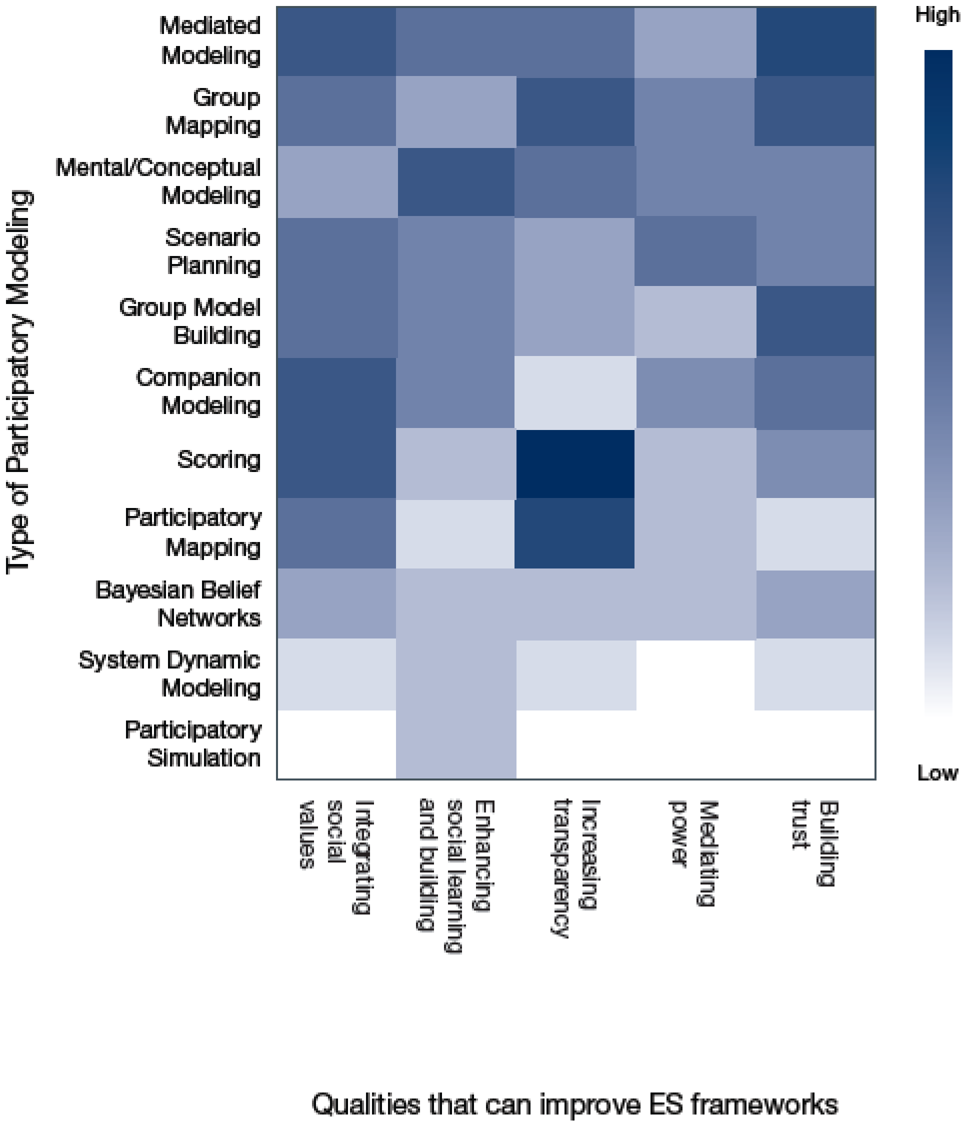



**Figure 5.5**: An assessment of how likely different types of participatory modelling are to contribute to the development of ecosystem service assessments in complex situations (Modified from Davies et al., 2015).

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, or both. Ideally, modelling approaches should consider both supply and demand, but it is difficult to define ecosystem service supply and demand in a consistent fashion due to variation in time scales, user groups and use patterns. There is also divergence between models that quantify ecosystem services and those that attempt some type of economic valuation, and whether they link benefits to specific beneficiaries. Linking benefits to different groups of beneficiaries is possible in many models, but not frequently done in practice. It is difficult to explicitly represent these links, since sub-groups often vary in their response to changes in ecosystem services. For example, Daw et al. (2015) showed that, while conservation appeared to improved fisheries ecosystem services, it actually reduced the ecosystem services available to some fishers if beneficiaries were disaggregated. Since links to beneficiaries can be extremely important, ecosystem service assessments need to develop this capacity.

Depending on the needs of a decision context, different modelling approaches are better suited to the task. To date, there has been relatively little cross-comparison of ecosystem service models (for an exception, see Bagstad et al. (2013b)). Additionally, there have not been significant attempts to develop models to guide ecosystem service policy implementation. This gap is not surprising due to the recent rise of interest in ecosystem service policies. Finally, while existing models have been used for agenda setting, and participatory modelling has been used to assess ecosystem services, there has not been much effort to guide the development of different types of approaches to ecosystem service modelling.

**Table 5.5**: Development of ecosystem service models by policy phase.

| Policy cycle phase | Agenda setting | Design | Implementation | Learning |
|---|---|---|---|---|
| Model type | Models for dialogue | Models for decision | Tactical models | Models for learning |
| Attributes | Articulating and bridging different perspectives | Defining and evaluating alternative policies/strategies | Identify strategies to enact policies | Evaluating, synthesizing, and creating models |
| Status of models | Little developed for ES | Most developed (e.g., InVEST, ARIES, EwE) | Little developed | Little developed |
| Approaches | Participatory modelling; adaptive management | See Figure 5.6 | Requires detailed local information, generalization difficult | Model comparison and integration (see chapters 6 and 8) |

Below we present a decision tree (Figure 5.6) that aligns the modelling approaches and tools presented here with the decision context presented in Chapter 2, using dimensions of spatial scale, and whether the model is spatially explicit; temporal scale (snapshot/single decision versus dynamic/sequential decisions); and whether the model includes valuation. It is recognised that additional dimensions will be involved in model selection (e.g. Chapter 2, Table 2.1). Figure 5.6 shows that there are more models and approaches at the regional scale than at the global scale, and that fewer of the models and approaches are dynamic in time or explicit in space.

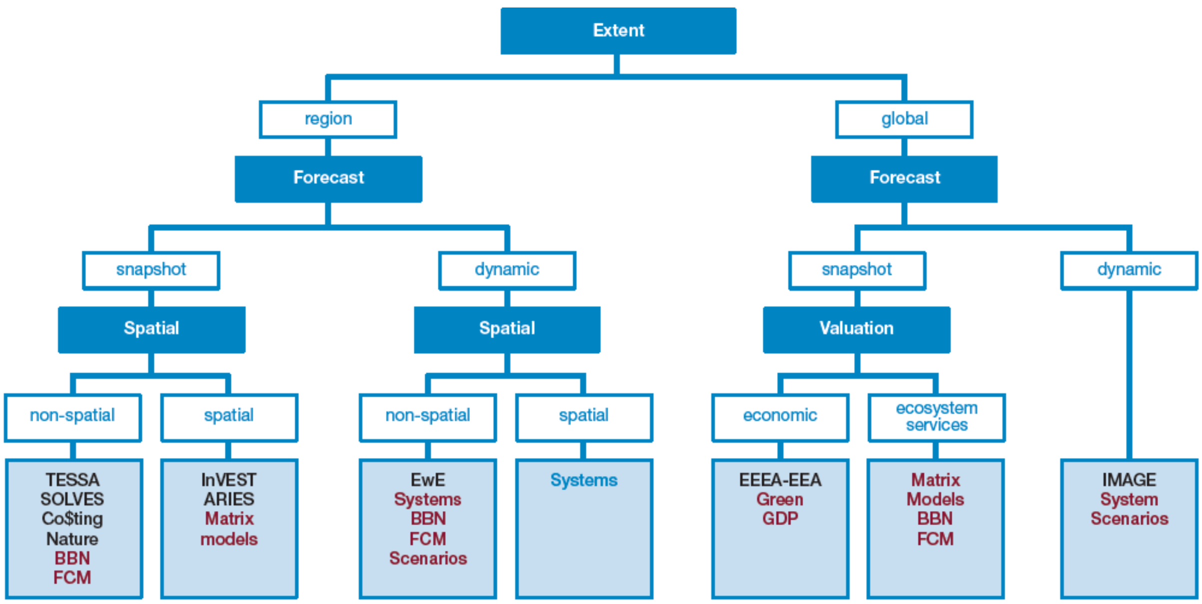


**Figure 5.6:** The decision tree outlines the sets of ecosystem service models that are currently available. The tree is defined by the extent of the model, type of forecast desired (snapshot or dynamic interaction among variables), and whether spatial or non-spatial analysis is conducted. The boundaries between different modelling approaches are less sharp than shown in this tree, because with some effort space or time can be incorporated in snapshot or non-spatial models. Modelling approaches are shown in blue, and modelling tools are shown in red.

There are currently no standardised modelling tools that assess how interactions among anthropogenic assets, institutions and biodiversity produce multiple ecosystem services.

Furthermore, the tools that do exist to explore the dynamics of ecosystem services are limited in the types of ecosystem services that they can assess. While there are a variety of tools that allow the assessment of marginal changes in landscapes on ecosystem services, there is not one single modelling approach that IPBES can adopt for its regional and global assessments of ecosystem services. It is unlikely that one modelling tool can be developed to forecast ecosystem services across a variety of social and ecological models, but efforts to standardise, share and cross-validate models and modelling approaches, combined with research on how to integrate models, would accelerate and improve the capacity of ecosystem service models at regional scales.

### 5.5.1 Methods for assessing and communicating uncertainty

Ecosystem service modelling to support decision making can be confounded by linguistic, stochastic and scientific uncertainty (defined in Chapter 1). Linguistic uncertainty in modelling arises when model parameters and variables are poorly defined or vague in their description. The development of a community of practice that clarifies and creates a shared language around key concepts and terms is essential to reduce this type of uncertainty. Specifically, it can be reduced with model metadata, tutorials, and accessible published examples of model architecture and applications. Stochastic uncertainty (i.e. system variability) is the natural variation in a system, which can be quantified through probability distributions, for example produced by Monte Carlo simulations of ecosystem service models, which can then be used to derive confidence intervals and risk profiles. Understanding this type of certainty is helped by access to historical data on key variables, for example how common have extreme rainfall events been, as well as forecasts of how those drivers are likely to change in the future. Scientific uncertainty describes the lack of complete knowledge in a modelled system and its parameters – 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. The impact of scientific uncertainty on policy choices can be reduced by using multiple models to screen and develop policies, as well as planning for surprises. Adaptive management has largely been developed to improve management in the face of all these sources of uncertainty (Walters, 1986).

Bark et al. (2013) make it clear that much stochastic and scientific uncertainty exists in ecosystem service assessments because of the complex physical and ecological systems that underpin the supply of ecosystem services, plus the large uncertainty inherent in socio-economic systems that value or demand ecosystem services. Schulp et al. (2014b) document five sources of uncertainty in ecosystem services quantified across Europe, including 1) indicators, definitions and frameworks that classify ecosystem services; 2) the level of process understanding, which leads to different quantification methods; 3) purposes of quantification that influence the selection of indicators; 4) biophysical and socio-economic input data; and 5) models used to quantify ecosystem services.

The robust communication of uncertainty has long occupied the Intergovernmental Panel on Climate Change (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 judgment, determined through evaluation of evidence and model agreement. It also used the quantitative reporting of uncertainty that stems from statistical or modelling analyses, expert opinion or other quantitative analyses. This describes uncertainty using a likelihood scale to express a probabilistic estimate of the occurrence of a single event or outcome. A system building upon the IPCC system of describing uncertainties and confidence could be developed for modelling ecosystem services and human well-being trajectories, scenarios and forecasts.

## 5.5.2 Data needs for model calibration, evaluation and development

### 5.5.2.1 Data availability

Model use and development is enhanced by the availability of data. Data availability requires both that the data exist, and that they are discoverable, accessible and usable by model developers and users. Data availability improves modelling capacity by enabling the use of more types of models, increasing the possibility of having available a model that can address a problem. Data availability also decreases the difficulty of using a model, and allows models to be more easily calibrated, tested, compared and evaluated (see Chapter 8). Furthermore, data availability is essential to the creation of new types of models.

Spatial data is particularly important for both modelling ecosystem services in general and IPBES assessments in particular. In particular, land-use and land-cover data are widely used by current ecosystem service models. Other types of data that are also used by ecosystem models include ecological, land-use, political, social, infrastructure and economic data. Ecological data include maps of species presence, vegetation communities, soil type, water, topography and geology. Social data include both land information such as maps of land use, land cover and land management, as well as political data such as political boundaries at multiple scales, demographic data such as age and gender, and other social data such as health, well-being and institutional membership. Useful economic data include land values, agricultural production, tourism and recreation, wild harvest data and estimates of non-consumptive use values. Institutional and cultural data are likely to grow in importance for models, but are currently not widely used. However, sharing indigenous and local knowledge is often more ethically complex than sharing geophysical data and requires different and more participatory approaches to developing databases, as this type of knowledge is often embedded in knowledge systems that are in political conflict and monitoring by colonial masters.

Open, free access to data has greatly accelerated model calibration, evaluation and development. Typically, models require the synthesis of data from a variety of sources, and having to pay for data can block the discovery of data. Data needs are widely shared across modelling tools, so that free open access to data needed to define and drive ecosystem service models and scenarios has the potential to increase the ability of people to use multiple tools, better compare them, and create useful model analyses and scenarios. Data that are available in formats that suit particular model inputs or are easily convertible into such formats further increase data accessibility.

Ecosystem service models also produce data that can be useful in modelling. For example, many ecosystem services have been mapped across the European Union (EU) Member States, and the US EPA is mapping ecosystem services in the USA with its EnviroAtlas. These and other types of model-produced data are useful to test, compare and improve other ecosystem service models.

Open access to data allows models to build on one another by constructing new types of synthetic data, which can then be used by other modelling projects to construct needed datasets. There are many examples of model-synthesised open source data that are widely used in global change research, and which include data on land use and land cover as well as climate data. Developing mechanisms for sharing data among models is also vital for better linking multiple models and better linking models and qualitative narratives in scenario planning (see Chapter 6). Data availability and sharing is also essential to enable more testing and the cross-validation of ecosystem service models.

Open data portals that host multiple datasets make it easier for modellers to discover and use data. Several data portals are currently provided by the UN and other organisations (e.g. the Food and Agriculture Organization (FAO) and the World Resources Institute (WRI)), as well as global change organisations such as the Centre for International Earth Science Information Network (CIESIN). Further improving the accessibility, interconnection and metadata of data related to ecosystem service models and scenarios increases the ease with which models can be created. Some ecosystem service models require specific types of data and have developed resources for sharing that data. For example, EwE requires food web data, and the community of practice using these models has developed databases of food webs that build on open databases produced by the Consultative Group for International Agricultural Research (CGIAR) fisheries researchers, as well as databases of models. Researchers are able to use these databases to develop and compare models and results. The comparison, development and use of ecosystem service models would be accelerated by making data for models, model output and the models themselves easily accessible.

#### 5.5.2.2 IPBES and data

The global unevenness of data and the uneven focus on ecosystem service models are two issues that are particularly important for IPBES.

Data availability is very uneven at the global level, and while some countries such as the USA have excellent open access to high-quality social and ecological data, in many regions of the world large amounts of data are not easily available. Data are currently most often available for high income countries and at the global scale. Because local and regional models use similar data but aggregated at different scales, their evaluation, use and development would be enhanced if databases were developed to support IPBES regional assessments, especially with data that are useful for assessing ecosystem services from indigenous and local perspectives, which may not be otherwise accessible. IPBES could enhance modelling capacity in developing regions and at regional scales by working to enhance access to data as well as developing libraries of models and semi-automated models set up to connect to available data. One approach is the use of web data sharing portals, such as the ESP spatial data mapping and sharing tool jointly developed by the European Commission Joint Research Centre and CSIRO (see http://esp-mapping.net/Home/). This ESP tool allows users to upload and download spatial data on mapped ecosystem services and query the database on the data available for different ecosystem services and locations.

Data are primarily available for ecosystem services that are closely connected to land use. Current models focus primarily on provisioning ecosystem services and carbon- and water-related regulating services. There is also some focus on tourism or other recreation-related cultural services. There is less of a focus on other ecosystem services. For example, more locally-important provisioning services, non-water- or non-carbon-related regulating services, and most cultural services, are neglected. These services require different types of data, and a better understanding of how biological features and society interact to produce these services.

Developing databases to support the modelling of a broad range of ecosystem services is necessary to understand the variation in ecosystem services in different locations, and to enable the creation of models that work well with indigenous and local knowledge.

## 5.5.3 Knowledge needs for model development and for ongoing evaluation and calibration

### 5.5.3.1 Sharing knowledge for model development

A key strength of modelling tools is their ability to be adapted to new contexts. Using a modelling tool in a new context requires a flexible model and some understanding of that context, but perhaps most importantly it requires being able to understand how to modify and use the modelling tool. Adapting a tool therefore depends on how difficult it is to use and how easy it is to learn to use it. Even a potentially easy-to-use tool is not useful if there is no documentation or training on how to use it. Understanding how to use a tool is usually greatly facilitated by a community of practice around the tool. In general, models that are easier to use have larger communities of practice (Figure 5.7). Only two of the approaches presented in this chapter have substantial communities of practice: InVEST and EwE (Figure 5.7). Both these modelling tools are moderately complex to use and have quite different strengths and weaknesses. EwE focuses on fishing-related ecosystem services – primarily non-spatially – but with a strong focus on dynamics and different beneficiaries, whereas InVEST is a set of interrelated models that spatially assess a broader range of individual ecosystem services, but it is not dynamic and not easily linked to multiple beneficiaries.

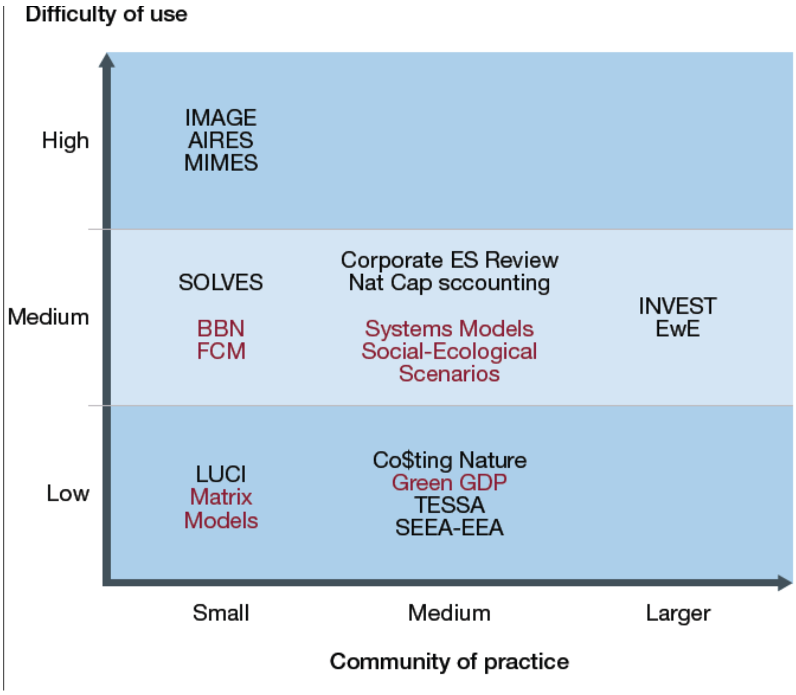


**Figure 5.7**: Comparison of the difficulty of use and community of practice existing for different modelling approaches (in blue) and frameworks (in red).

#### 5.5.3.2 Developing new knowledge for model development

Following the MA, Carpenter et al. (2009) laid out a number of research challenges for ecosystem service science, which included developing models that address key social challenges, developing the ability to forecast ecosystem services, modelling trade-offs among ecosystem services, being able to address non-linear and abrupt change, and better modelling the diverse interactions among people and nature and how ecosystem services interact with other factors to influence human well-being. While substantial progress has been made in developing new types of applied models and assessing trade-offs among ecosystem services, most of the other challenges remain. Two of Future Earth's research programmes, ecoServices and Programme on Ecosystem Change and Society (PECS), which are strongly related to modelling and scenarios of ecosystem services, have also identified similar issues.

Future Earth's ecoService research programme aims to improve the incorporation of ecosystem service research into decision making for the sustainable use of natural resources to improve human well-being by addressing three main research questions: i) how are ecosystem services co-produced by social-ecological systems, ii) who benefits from the provision of ecosystem services, and iii) what are the best practices for the governance of ecosystem services (Bennett et al., 2015)? PECS explores four key areas for improving ecosystem service science: 1) improving the understanding and governance of social-ecological interactions between regions; 2) better understanding long-term drivers of social-ecological change; 3) exploring how power relations, justice and ecosystem stewardship interact; and 4) investigating how to better connect a diverse ecosystem service science to society (Fischer et al., 2015).

The research priorities identified by these research programmes align fairly well with the gaps we identify in this chapter. However as discussed above, there is a need to develop models for agenda setting and learning, as well as to further develop models for learning that enable model testing, comparison and verification (Table 5.5). Additionally, more strongly incorporating stakeholders, as well as indigenous and local knowledge, in IPBES assessments will require using and developing models for dialogue. Addressing these research topics should significantly enhance the capacity to understand the production and dynamics of ecosystem services, and it would benefit IPBES if the issues and challenges that are identified in IPBES assessments and syntheses could be shared with these research programmes to ensure that they address these research questions in ways that enhance the capacity of IPBES. We provide more detail on four particular knowledge needs that are important to IPBES: linking biodiversity and ecosystem services, linking ecosystem services to human well-being, enhancing model transparency and accessibility, and using and integrating multiple models.

#### 5.5.3.3 Linking biodiversity and ecosystem services

Rapid development has taken place in both models of biodiversity and ecosystem services over the past decade. However, these models are only weakly connected to one another, and the research communities working on biodiversity and ecosystem services are also not very connected to one another. Research on links between biodiversity and ecosystem function is one way in which connections have been strengthened between the two communities, but the accessibility of land-cover, land-use and other spatial data compared to functional biodiversity data has meant model development has focused on these variables as proxies for biodiversity and biodiversity change.

There is a substantial opportunity for linking, bridging and synthesising the types of models discussed in Chapter 4 with those in Chapter 5; however, because ecosystem services are co-produced by nature, institutions and anthropogenic assets these linkages require more than just using the outputs of biodiversity models as inputs into ecosystem service models.

#### 5.5.3.4 Linking to human well-being

Most ecosystem model development to date has focused on assessing ecosystem services, with minimal attention given to how these ecosystem services link to human well-being, especially that of diverse groups of beneficiaries. Modelling the impact of ecological changes on human well-being is not well developed, partially because our understanding of human well-being is poor, but also due to the lack of involvement of human well-being researchers in ecosystem service modelling. However, some recent advances have been made. Increasing evidence demonstrates that contact with nature provides many physical and mental health benefits (Hartig et al., 2014; Bauch et al., 2015; Bratman et al., 2015; sCBD and WHO, 2015; Townsend et al., 2015; Wheeler et al., 2015; Whitmee et al., 2013). Access to parks and green spaces encourages increased physical activity, and being close to parks and nature can reduce depression, anxiety and other mental health problems. Recent groundbreaking research has also shown that brief nature experiences reduce neural activity in a part of the brain associated with a heightened risk of mental illness (Bratman et al., 2015).

The human health benefits of experiencing nature are especially important in richer countries, given the large number of people with sedentary lifestyles and associated increases in Western lifestyle diseases. For example, it is estimated that 56% of Australians have a sedentary lifestyle with very low levels of exercise. An inactive lifestyle greatly increases the risk of a heart attack, stroke, type-2 diabetes, cancer and osteoporosis, which together are estimated to cost the Australian economy about $13.8 billion per year, equating to about $1,660 per inactive person (Medibank Private, 2008). A recent study estimated significant health (including 2,000 fewer deaths and 6,000 fewer incident cases of disease) and economic (including $96m health sector cost savings and a gain in 114,000 working days) benefits to the Australian economy given a 10% reduction in the population’s physical inactivity (Cadilhac et al., 2011).

Developing tools that better link human well-being and ecosystem services will require investment and the transdisciplinary collaboration of policymakers with natural and social scientists to develop new frameworks, methods and tools.

Most modelling tools have been developed to aid decision making in situations that are clearly defined and not contested (Figure 5.5), and there is a need to develop tools that work in other types of decision contexts. Of particular relevance to IPBES are tools that allow bridging across knowledge systems and that allow indigenous and local knowledge to be included. Particular issues that need more model development include 1) assessing the impact of ecological change on different groups of people, 2) incorporating different knowledge systems in modelling operation and practice, 3) considering the co-production of ecosystem services as well as the spatial distribution of services and beneficiaries, 4) adapting model communication for different decision contexts, 5) better incorporating social-ecological feedbacks in models, and 6) developing methods for better integrated ‘soft systems’ approaches with quantitative spatial models.

#### 5.5.3.5 Model transparency and accessibility

For modelling approaches and tools to be widely used and trusted they should be available at no cost through open access distribution.

Even a minimal cost can prevent people from assessing the utility of the tool, learning how to use it, or evaluating its performance. Some modelling tools, such as Vensim, offer a version that is 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 is 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 a 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.

Both InVest and EwE have taken this approach for their modelling tools. An example of an open approach taken by a modelling community is the openABM project (openABM.org), which provides an excellent example of a general model database. Some journals publishing agent-based models (ABMs) require that papers 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, adapt it, or integrate it into a more complex model. Such practices could be an effective part of IPBES capacity-building activities.

#### 5.5.3.6 Using and integrating multiple models

A complex decision context does not necessarily require a complex model. An increase in the number of variables explicitly modelled exponentially increases the complexity of a model, because each additional variable added to a model requires representing how that variable is connected to existing variables thereby greatly increasing the difficulty of creating, parameterising, applying, analysing and communicating a model.

In complex decision contexts, complexity can often be addressed 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 potentially 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. Figure 5.4 highlights that different phases of the policy process require models with different strengths, which suggests that IPBES assessments should consider having a toolbox of models that they use rather than attempting to identify a single model or modelling approach that is appropriate to all contexts.

## 5.6 Capacity-building needs

The capacity of IPBES stakeholders (e.g. from local and indigenous communities, scientific communities, civil society, industries and governments) to develop, use or analyse models or scenarios of ecosystem services is greatly limited by geographical unevenness in 1) the development of ecosystem services models and scenarios science, 2) access to relevant and quality databases, 3) methods for integrating multiple knowledge systems, 4) the availability of funding for such activities and 5) access to training on the use and implementation of available tools and methodologies, and 6) communities of practice that can provide support and access to modelling tools and techniques. See Chapter 7 for a more detailed discussion of these issues.

To address the current limitations, there is a great need to:

1) build communities of practice and forums (i.e. partnerships and networks) that build upon the success and lessons learned for existing communities of practice around ecosystem service tools such as InVEST and EwE, as well as other tools such as Marxan and organisations such as the Ecosystem Service Partnership and The Economics of Ecosystems and Biodiversity (TEEB);
2) develop accessible standards and documentation for models and scenario-building tools;
3) standardise and organise useful datasets, models and scenario building across various scales, such as EcoBase which has been developed to share EwE models (Colléter et al., 2013);
4) create a dynamically updated catalogue of models and scenario-building tools that includes an evaluation of how they can fit in with different decision contexts and phases of the policy cycle;
5) develop practical transdisciplinary methods for bridging multiple knowledge systems (Tengö et al., 2014) in modelling and scenario building to enable the production of more legitimate, robust and inclusive policy recommendations and outcomes; and
6) improve access to development, training and the use or applications of model and scenario tools for policymaking, in particular by developing strategies to improve the ability to develop, use and analyse these tools among indigenous and local knowledge holders, as well as among researchers in countries that lack ecosystem service assessments.

IPBES assessments have the potential to help catalyse the development of global communities of practice for ecosystem service modelling and scenario analysis (see Chapter 7 for a detailed discussion), but achieving this goal will require new approaches to the design and operation of these assessments.

## 5.7 Summary and synthesis

This chapter offers an assessment of the rapidly changing landscape of methods for assessing and forecasting the benefits that people receive from nature, and how these benefits are shaped by institutions and various anthropogenic assets. There has been an explosion of activity in understanding and modelling the benefits that people receive from nature, and this explosion has produced a diversity of approaches that are both complementary and contradictory. However, there remain major gaps in what current models can do. For example, they are not well suited to estimating most types of benefits at national, regional or global scales. They are focused on decision analysis, but have not focused on implementation, learning or dialogue. This gap in particular means that current models are not well suited to bridging multiple knowledge systems; however, preliminary efforts are being made to achieve this and there appears to be a clear demand for this type of activity. Furthermore, while participatory social-ecological scenarios are able to bridge multiple knowledge systems in their assessment and analysis of multiple ecosystem services, the social-ecological scenarios community is fragmented. Consequently, IPBES has an excellent knowledge base to build upon, but a real investment in building a more integrated modelling and scenarios community of practice is needed to produce a more complete and useful toolbox of approaches to meet the needs of IPBES assessments and other assessments of nature's benefits.

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# 6 Linking and harmonising scenarios and models across scales and domains

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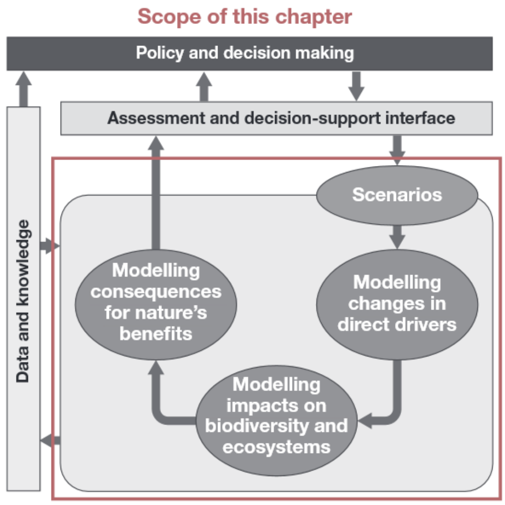
**Purpose of this chapter:** Critically reviews approaches to linking and harmonising the various types of scenarios and models described in Chapters 3, 4 and 5 across scales, domains and elements, thereby better serving the diverse needs of policy and decision making (as covered in Chapter 2); proposes ways in which IPBES might best achieve such integration in its own assessments.

**Target audience:** Aimed mostly at a more technical audience such as scientists and practitioners wanting to identify appropriate approaches to linking and harmonising scenarios and models for different applications.




## Key findings

**Linking models and scenarios can be used to aid understanding of the positive and negative impacts of an action across interconnected scales (time, space, social organisation) and elements (biodiversity, ecosystem services, human well-being) (Sections 6.1, 6.5). However, linking models and scenarios is not appropriate in every decision context, particularly when error propagation increases uncertainty to an unacceptable level.** Existing families of approaches include one-way (information is passed in one direction between two or more models), two-way (information is passed in both directions between models allowing for feedbacks), loose coupling (meaning that model output can be computed separately) and tight coupling (integrated, requiring the simultaneous processing of multiple models) (Section 6.2). One-way loose coupling (quantitative and/or qualitative) is used the most frequently because it is relatively straightforward and often meets the desired objectives. Two-way coupling is more complex, but necessary and beneficial in some situations to explore and capture feedbacks.

**Harmonisation enables comparison across models and scenarios, which is a necessary step to understand the uncertainty around associated with possible outcomes (Section 6.4).** It also involves upscaling and downscaling models and scenarios in organisation, space and time, as well as model benchmarking. Upscaling and downscaling along a social organisational scale requires an awareness of humanly- imposed boundaries and conventions (Section 6.4.1). The ecosystem services approach can be considered an organising principle for harmonisation along an organisational scale (Section 6.4.1.1). Spatial downscaling provides information for local-scale policy making when high resolution information is not available. Statistical downscaling is most often used; however, dynamic downscaling that is based on mechanistic models may be more appropriate than statistical downscaling in systems where the relationship between coarse- scale and fine- scale dynamics are complex and non-linear, or where observational data are insufficient (Section 6.4.1.2). In the process of upscaling, quantitative approaches to preserve the quality of the original information should be applied whenever possible; otherwise, it the upscaling can contribute to scaling uncertainties (Section 6.5.2). Upscaling and downscaling methods across temporal scales are in principle similar to those for across spatial scales (Section 6.4.1.3).

**Multi-scale scenarios that link global and regional-scale scenarios have been useful in informing environmental assessments that need to consider drivers at different scales (Section 6.4).** Approaches for developing multi-scale scenarios include using global-scale scenarios as boundary conditions for regional-scale scenarios, translating global-scale storylines into regional storylines, using standardised scenario families to independently develop scenarios across scales, and using global scenarios directly for regional policy contexts. However, there are few approaches and examples for upscaling regional scenarios for global assessments, and few examples (Section 6.4.2).

**Multi-model benchmarking using species-level biodiversity models at the species level or ecosystem services models is not available.** Benchmarking is the process of systematically comparing sets of model predictions against measured data to evaluate model performance. It also helps identify processes that may be poorly represented in models (Section 6.4.3).

**Uncertainties in different biodiversity and ecosystem services models that are linked across spatial and temporal scales, elements and domains may potentially propagate through the chains of models, affecting the ultimate envelope of uncertainty.** Available options to address errors associated with linking models include not linking the models, limiting the extent of model linkages, and exploring the envelope of uncertainty resulting from model linkages. When system processes interact across scales, resulting in non-linear dynamics, harmonising models and their outputs across these scales is more likely to result in scaling error. In such cases, the use of multiple scale models perform produces a better result than the use of single scale models (Section 6.5).

# Key recommendations

**The Intergovernmental Platform on Biodiversity and Ecosystem Services (IPBES) Task Force on Capacity Building could help to foster the development of communities of multi-disciplinary researchers and practitioners to harmonise and link across models, scales, domains and elements (Section 6.1).** This would encourage shared learning from experience gained from different approaches employed in different parts of the world – for example across different regions or countries.

**The IPBES Global and Regional Assessments would benefit greatly from not limiting their work to a particular scale, but rather use using multi-scale scenarios (Section 6.4) that are coupled both loosely and tightly (Section 6.1).** The loose-coupling approach is particularly suitable for framing stakeholder issues, while the tight-coupling approach also allows the consideration of feedbacks among between scales, elements and domains and promotes a more detailed system understanding.

**The IPBES Task Force on Knowledge, Information and Data could work with the scientific community to define a set of standard conditions and components for 'IPBES-compatible' model and scenario components that share common ground (Section 6.4, and also Chapters 5 and 8).** This could be similar to the approach that has been successfully implemented through coordinated efforts between the Intergovernmental Panel on Climate Change (IPCC) and the scientific community.

**The IPBES Task Force on Knowledge and Data could play an important role in encouraging the incorporation of ecological processes (e.g. population dynamics or the biogeography of groups of animals) into integrated assessment models (IAMs) (Section 6.3).** This would allow these classes of models to address a broader range of questions related to biodiversity and ecosystem services.

**The IPBES Global and Regional Assessments should consider exploring the use of existing scenario archetypes (families) to link and harmonise scenarios that best respond to their questions.** Common scenario families include economic optimism, reformed markets, global sustainable development, regional competition, regional sustainable development and business-as-usual (Section 6.4).

**To improve the linking and harmonising of models and scenarios for assessing ecosystem services, human well-being and policy options, the IPBES Task Force on Knowledge, Information and Data could facilitate the development of an open source data infrastructure to share multi-disciplinary data, toolkits and tested methods, and to promote the use of common terminology (Section 6.4).** This would allow the informed linking and harmonisation of scenarios and models, as well as model benchmarking.

## 6.1 Importance of linking and harmonising models and scenarios

### 6.1.1 Introduction

Models and scenarios are important tools for understanding and communicate communicating the effects of natural and human drivers on biodiversity and ecosystem services (Chapters 3, 4 and 5). The temporal, spatial, and social organisational scales that a single modelling or scenario assessment focuses on are generally specific to particular policy contexts (Chapter 2).

However, biodiversity, ecosystem services and their drivers are interconnected, and can span multiple spatial and temporal scales, domains and elements of the IPBES framework (see Chapter 1 and Glossary). Thus, linking models or scenarios and harmonising across different scales, domains and elements are important steps in advancing our understanding of how we can sustain human well-being while ensuring the conservation of biodiversity (Steffen et al., 2015; Mace et al., 2012; Dearing et al., 2014).

Here, the concept of 'domain' includes the dimensions of space and time, disciplines and knowledge.

Overall, this chapter aims to: a) summarise existing approaches and initiatives that link and harmonise models and scenarios across scales, domains and elements; b) discuss relevance to policymaking; c) identify knowledge gaps; and d) propose possible ways for IPBES to undertake multi-scale/domain/element linkages and harmonisation to assess biodiversity and ecosystem services. This chapter builds on Chapters 2 to 5 to assess the availability of tools and methods for linking and harmonising scenarios and models of drivers of biodiversity (Chapter 3) and to assess the impacts of these drivers on biodiversity, ecosystem functions (Chapter 4) and benefits to people (Chapter 5) to inform policymaking at specific spatial and temporal scales (Chapter 2). Models for biodiversity and ecosystem services can be run at a wide range of spatial, temporal and organisational scales, depending on the elements, domains and processes that they represent (Figure 6.1, see Section 6.2). Here, we focus on both short (10–15 years) and long (multi-decadal) temporal scales, and on national, regional and global (*sensu* IPBES) spatial scales. We present case studies selected across a variety of elements and applications to showcase approaches that tackle complex issues.

### 6.1.2 Linking and harmonising models and scenarios: why and why not

The linking of models and scenarios can be used to aid understanding of the positive and negative impacts of an action across interconnected elements, by revealing the interactions and feedbacks across multiple elements and domains of social, economic and natural systems (Carpenter et al., 2006).

Decision makers, from individuals to global institutions, are unlikely to have knowledge about all the impacts of their chosen actions within an element and across multiple, interconnected elements (Chapter 2). An action may impact individual elements in different and often unexpected ways across spatial and temporal scales, as well as potentially affect multiple elements. For example, damming a river impacts fish upstream and downstream of the dam (migration barrier; *spatial impacts*), immediately and in the longer term (altered water flow, sediment accumulation in reservoir; *temporal impacts*), and impacts fish, aquatic and terrestrial plants, and people (*multiple elements/domains*).

For some decision contexts, multiple models and scenarios exist, or could be developed, that provide different information for decision makers (Chapter 2). Models or scenarios may differ because they a) were developed to address subtly different questions for different audiences (e.g. composition and function of biodiversity, temperature and precipitation for climate) and therefore produce different outputs (e.g. carbon ecosystem service models may output carbon stocks or carbon sequestration); b) use different input data (e.g. different biophysical layers for species distribution modelling); c) represent different components/elements within the model/scenario (e.g. biodiversity models may incorporate metabolism, reproduction, growth, dispersal, or mortality); d) use different methodologies or techniques (e.g. from correlative to process-based models, see Chapter 4); or e) cover different spatial and/or temporal scales.

To bring models or scenarios together and compare them, they need to be made compatible or consistent with one another; this process is referred to as 'harmonisation'.

Harmonisation is related to the concept of interoperability, or the ability of different information technology components, systems and software applications to communicate and exchange data accurately, effectively and consistently, and to use the information that has been exchanged (Heubusch, 2006; Matott et al., 2008; Laniak et al., 2013). Models and scenarios can be harmonised in multiple ways, by using standardised inputs (e.g. all IAMs used in the IPCC Fifth Assessment Report use the same harmonised land-use data, Hurtt et al. (2011), }), by using agreed output metrics, evaluation or benchmarking against common observational data sets (e.g. global circulation models to be included in IPCC reports need to be able to hindcast historical temperature trends, derived from multiple sources), or by specifying the key components and elements that need to be represented in the model or scenario.

Linking multiple models and scenarios is not appropriate in every decision context.

This is for a number of reasons. First of all, each model and scenario comes with its own assumptions. When these assumptions are incompatible, linking the models/scenarios produces an uninformative output (Laniak et al., 2013). Secondly, the causality of links across elements is sometimes poorly understood. In such situations, the output of linked models/scenarios would become a poor representation of phenomena. Thirdly, model output-input chains and feedbacks are often complex, difficult to debug and potentially result in error propagation and uncertainty. This becomes unhelpful for decision making when error propagation increases uncertainty to an unacceptable level (Dunford et al., 2014; see Section 6.5). Voinov and Shugart (2013) cautions that, in some cases, the software engineering approach of mechanically connecting models as software can result in conceptually ambiguous products or 'integronsters', which seem to be technically correct but make little sense as realistic system models and decision-support tools. In addition to data integration that checks consistency with model specifications (units and temporal-spatial scales) and assumptions when passing data between models, the semantic integration of concepts and assumptions is also important when linking models. Given the potential complexity in linking models and scenarios, the amount of linkage among models/scenarios needs to be tailored to the decision context (Chapter 2).

Scenarios and models may differ because they do not share the same values, or the same world views (i.e. decision uncertainty), and it may be important to present these differences clearly. Likewise, models may differ in the drivers and processes included. Without a clear understanding of these drivers and processes, it is not beneficial to harmonise the models and their outputs.

Standardising inputs and output metrics and components included in models is likely to reduce the uncertainty around estimates (e.g. by using standardised model inputs or by removing outliers). However, this usually results in models/scenarios that give more similar outputs which may be more precise, but not necessarily accurate and therefore less relevant to policy relevant. By a priori standardising inputs and components a priori as well as ensuring validation against a standard dataset, models/scenarios that are projecting low frequency/-high impact events may be excluded (Levin, 2003) (e.g. the 2008 financial crisis or abrupt climate change).

## 6.2 Approaches for linking and harmonising models and scenarios

Models or scenarios developed for different spatial and temporal scales, domains and elements (i.e. the elements of the IPBES framework, see also Figure 6.1) can be linked or harmonised using several approaches (Table 6.1). Linking takes place by feeding using the outputs of one model as input to another model, which can be done iteratively (two-way or tight coupling) or off-line (one-way or loose coupling). Models and scenarios can also be linked qualitatively (e.g. through narratives or description of storylines descriptions). Models and scenarios that describe different elements may also be combined quantitatively or qualitatively to provide a more holistic assessment, as done by Integrated Assessment Models (IAMs), and, more generally, integrated environmental modelling Laniak et al., 2013).

Linking takes place by using the outputs of one model as input to another model, which can be done iteratively (two-way or tight coupling) or off-line (one-way or loose coupling). Models and scenarios can also be linked qualitatively (e.g. through narratives or descriptive storylines).

Integrated environmental modelling broadly refers to modelling approaches that represent holistic system-level thinking by using quantitative and participatory methods for defining, selecting, integrating, and processing the combination of environmental, social and economic information needed to inform decisions and policies related to the environment (Laniak et al., 2013). Note that links are frequently being made across spatial and temporal scales, and elements may act on one another at different scales (also see IPBES/3/INF/4, http://ipbes.net/, Chapter 2). For example, historical global climate data may be used as an input for modelling the current distribution of species at the national scale, which is then used to estimate a provisioning ecosystem service (blue arrows in Figure 6.1).

**Table 6.1:** Summary of different approaches to linking and harmonising models and scenarios.

| | Approaches | Model | Scenario |
|---|---|---|---|
| Linking | Output-input, one-way coupling (section 6.3.1) | x | |
| | Output-input, two-way coupling (section 6.3.1) | x | |
| | Combining outputs qualitatively (section 6.3.2) | x | x |
| Harmonizing | Standardization of metrics (input and/or output) through a) classification schemes and taxonomies; b) converting across dimensions, domains and organizational levels; c) scaling in time and space (sections 6.4.1-2) | x | x |
| | Benchmarking (section 6.4.3) | x | |

The harmonisation of models and scenarios occurs across domains and spatial and temporal scales *within* an element.

In particular, harmonisation involves the standardisation of metrics (e.g. output metrics of models, conditions for scenarios for $CO_2$ concentration or agricultural production) and input data (e.g. land use, or temperature), or both.

For nominal variables, this can be achieved by adopting standard classification schemes (e.g. the unified classification of species threats and conservation actions, Salafsky et al., 2008).

Harmonisation often involves upscaling and downscaling models and scenarios in space and time, as well as model benchmarking.

Benchmarking is not applicable to scenarios because these are by definition alternatives to one another. Harmonised models and scenarios and their outputs facilitate model linking, error detection and uncertainty estimation, and ultimately decision making.

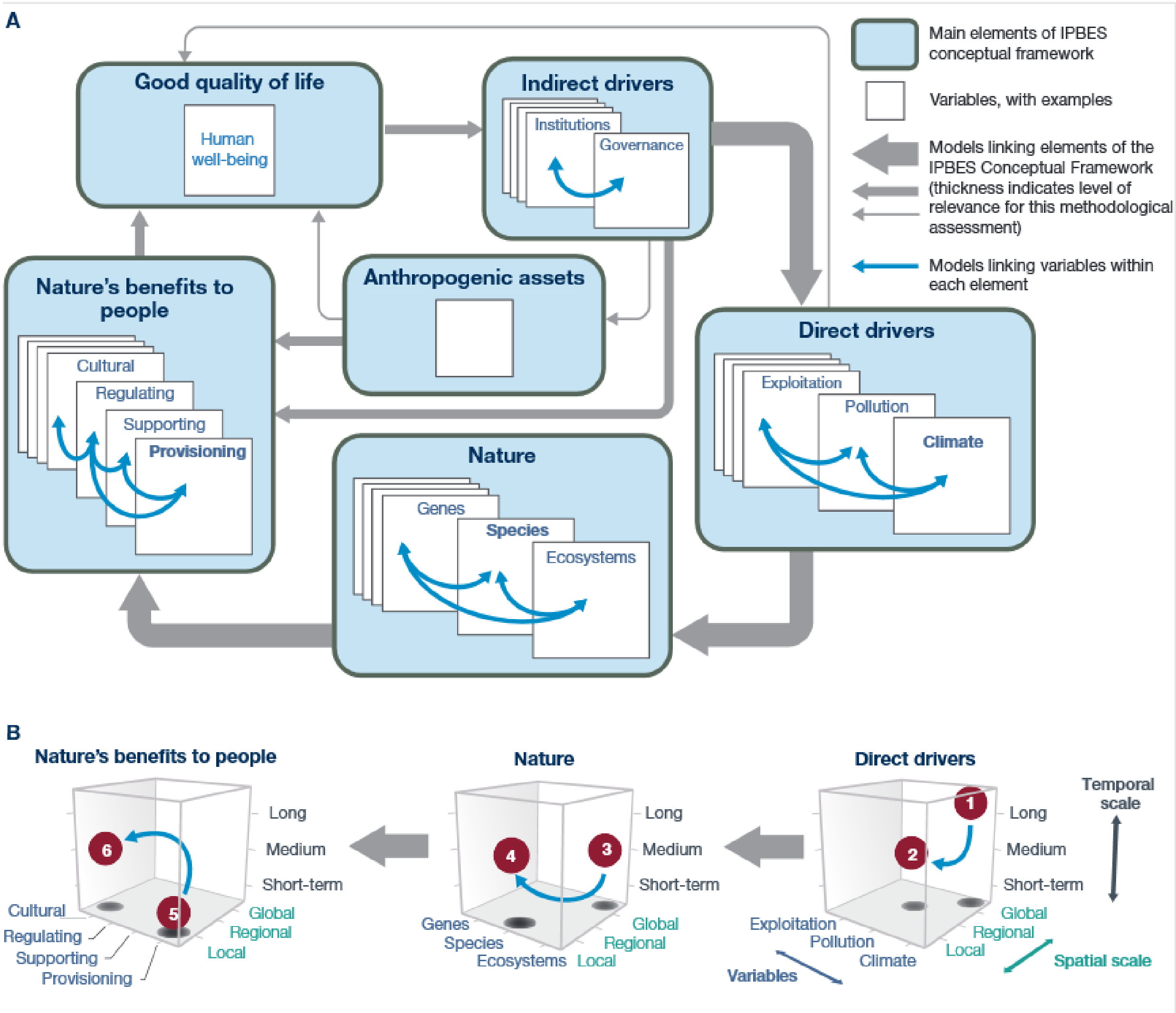



**Figure 6.1:** Linking models among the six elements of the IPBES conceptual framework, among variables (or organizational scales) within each element, and among spatial and temporal scales of each variable. Each element has multiple dimensions (Panel A) including temporal and spatial scales, and disciplinary and organizational domains. Blue arrow explained in text. Panel B provides illustrative examples of how linking and harmonizing models facilitates assessment of biodiversity and ecosystem services. For example, centennial-scale outputs from climate and ocean conditions from global-scale Earth System Models (1) can be used as inputs to project decadal and regional changes in level of marine contaminants e.g., methyl-mercury (2). Outputs from (1) and (2) can be used to project changes in regional marine ecosystems structure and functions (3), which can then be linked to

species-level models to assess the effects of these direct drivers on species abundance and diversity in different local areas (4). The projected potential distribution and productivity of living marine resources can be used to assess their benefits to local communities through fisheries (provisional service) (5). Through understanding how fisheries relate to traditions and culture e.g. through the use of indigenous and local knowledge, the results can also help assess the impacts of the direct drivers on local culture (6).

## 6.3 Linking models and scenarios of biodiversity and ecosystem services

### 6.3.1 Input-output model coupling

Models representing different components of the social-ecological system are linked through either one-way (offline) or two-way coupling (which allows feedback). In both cases, outputs from one model feed into another model as inputs. For example, in modelling the effects of changes in ocean conditions (temperature, primary productivity, oxygen level and acidity, and the resulting species range shifts) on marine ecosystems in the Northeast Pacific coastal area, Ainsworth et al. (2011) took simulated changes in ocean conditions from a coupled ocean-atmospheric earth system model and projected range shifts from species distribution models as inputs (forcing factors) into trophodynamic food web models to simulate the effects of multiple $CO_2$-related drivers on marine ecosystems and fisheries yields. Visconti et al. (2015) used models of climate and land-use change based on scenarios of socio-economic development as an input to species distribution models that projected distributions into the future, and assembled these projections into policy-relevant indicators of biodiversity change (Box 6.1). Two-way coupling includes feedbacks of inputs-outputs between models. For example, a marine ecosystem model, Atlantis (Fulton et al., 2011), links model components describing ocean biogeochemistry, the lower-trophic level ecosystem, the upper-trophic level ecosystem and human activities (with a focus on fishing), in which outputs from the components affect one another directly or indirectly over space and time.

**Box 6.1:** Using scenarios of global change to project species distributions and biodiversity trends into the future

To project trends of about 400 species of large terrestrial large mammals under scenarios of global change, Visconti et al. (2015) used multiple models (climate models, an integrated assessment model and species distribution models) linked using the output-input method with one-way loose coupling.

The impact of climate change on the geographic ranges of species was quantified by fitting bioclimatic envelope models to the present-day species' distributions, and projecting these under future climates associated with two scenarios of socio-economic development until 2050. The two scenarios, developed for the Rio+20 conference held in Rio in 2002, represent business-as-usual production and consumption patterns and rates, or reduced consumption (PBL, 2012). For each socio-economic scenario, three species responses to climate change were tested: 1) species cannot disperse into new climatically suitable areas; 2) species can expand their distributions each generation by a median dispersal distance estimated using statistical models; or 3) species adapt locally (their geographic ranges are not affected by climate change). Projected species ranges were further assessed for compatibility with the fine-scale ecological requirements of the species using habitat suitability models (Rondinini et al., 2011; Visconti et al., 2011) based on species' land-cover and altitudinal preferences, and sensitivity to human disturbance. These models were applied to projected land-use maps from the Integrated Model to Assess the Global Environment (IMAGE) (Bouwman et al., 2006) under each scenario, to quantify for each species the extent of suitable habitat for each species. The distribution projected under each climate change scenario was taken as the extent of occurrence. The extent of suitable habitat was treated as the maximum potential value of area of occupancy. The number of mature individuals of a species was estimated by multiplying the area of occupancy by the population density from observed and modelled data. These parameters were applied to Red List criteria to evaluate each species' Red List category for each year under each scenario, from which the overall Red List Index (RLI) was calculated following Butchart et al. (2007) (Figure Box 6.1). The uncertainty around the proportion of mature individuals and proportion of suitable habitat occupied (area of occupancy/extent of suitable habitat) was incorporated into RLI projections by randomly sampling these parameters from a distribution with intervals gathered from the literature and by performing a Monte Carlo simulation. Estimates of mature individuals for each species and each year were used to generate the Living Planet Index (LPI) for each scenario following Collen et al. (2009). The methodology was validated through by hind-casting species distributions and biodiversity indicators from 1970.

Testing these on terrestrial carnivore and ungulate species, Visconti and colleagues found that both indicators decline steadily, and by 2050, under a business-as-usual scenario, the LPI declines by 18–35% while the extinction risk increases for 8-–23% of the species, depending on assumptions about species' responses to climate change. Business-as-usual will therefore fail Convention on Biological Diversity (CBD) target 12, which is to of improve the conservation status of known threatened species. An alternative sustainable development scenario reduces both the extinction risk and population losses compared with the business-as-usual scenario, and could lead to increases of mammal populations.

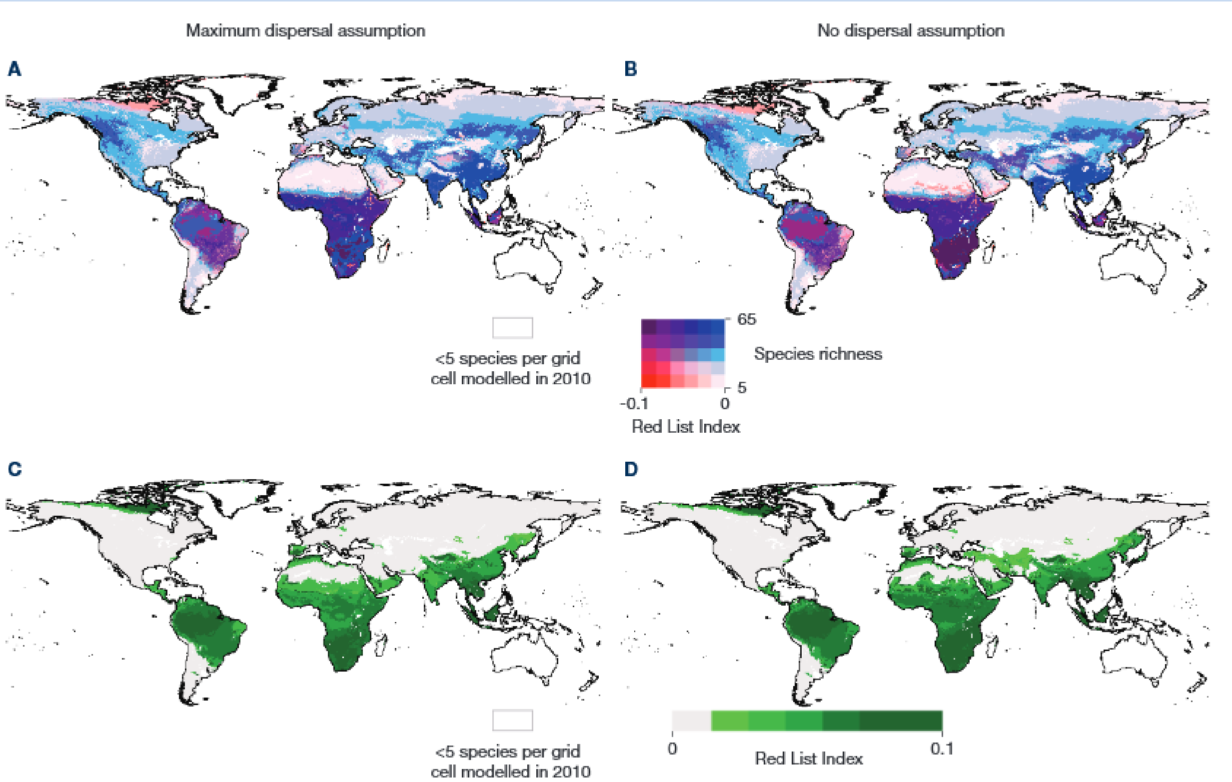



**Figure Box 6.1:** Spatial patterns of trends in Red List Index for mammalian carnivores and ungulates. (A,B) Spatial pattern in species richness and trends in the Red List Index (d-RLI) between 2010 and 2050 under a business-as-usual scenario, with land use and climate change and assuming maximum species dispersal (A) and no dispersal (B). The colour of each cell is a blend of species richness (blue tones) and difference in Red List Index between 2010 and 2050 (red tones). (C-D) Relative improvements in d-RLI for the reduced impact scenario relative to business-as-usual for year 2050 under maximum dispersal (C) and no dispersal (D). Areas in white (including Australia) contain fewer than 5 species of carnivores and ungulates per grid cell modelled in 2010. A negative difference of the Red List Index indicates an increase in the aggregate extinction risk of carnivores and ungulates (the average conservation status of species deteriorates), while a positive difference indicates a decrease in the aggregate extinction risk (the average conservation status of species improves). (Modified from Visconti et al. 2014, *Projecting Global Biodiversity Indicators under Future Development Scenarios.* Copyright © 2014 by John Wiley Sons, Inc. Reprinted by permission of John Wiley & Sons, Inc).

The choice of coupling methods depends on the dynamics of the modelled systems and the objectives of the models.

One-way coupling is simpler to implement than dynamic two-way coupling because the models can be run sequentially.

The responses of the modelled system are also more predictable, because feedbacks are not allowed (e.g. predicting changes in fish distribution and production driven by Earth system model outputs versus tightly-coupled system dynamic models, where changes in fish stocks feedback to changes in ocean biogeochemistry and climate). On the other hand, non-linear system dynamics and feedback between model domains cannot be directly revealed with models that are coupled one-way.

Two-way coupling is more realistic for understanding social-ecological systems where feedbacks and resulting non-linear responses are common among domains and elements.

However, this is technically more difficult, particularly if components operate at different temporal and spatial scales. The model responses are also less predictable and may result in large internal variability.

Most examples of impact models relevant to IPBES concern one-way coupling between models (Table 6.2). Two-way coupling or full socio-ecological systems modelling is not common. Rare examples include IAMs (e.g. IMAGE 3.0, Box 6.2), which only represent very general system characteristics and are of limited use for regional or local policymaking or stakeholders.

Integrated assessment models combine components (sub-models) representing the future development of human societies, including major sectors such as energy use, industrial development and land use, which are important for making projections about the future of human and natural ecosystems (Harfoot et al., 2014). Currently, the main applications of IAMs are modelling climate change and the effects of climate mitigation. In most IAMs, their sub-models – including both natural and human subsystems – are linked, although dynamic linkages are not commonly represented in most IAMs (Harfoot et al., 2014). An example of natural systems sub-models in an IAM is the linkage between hydrological models providing inputs regarding water and nutrient supply into terrestrial vegetation models. For human systems sub-models, examples include components representing the energy sectors that capture the supply and demand of energy as links to industrial development, population demand and commodity prices. There are also components that link natural-human systems such as food production, linking vegetation and land use with societal demand, energy sources (particularly from bioenergy crops) and commodity prices.

**Table 6.2**: Impact model types of special relevance for IPBES (adopted from Chapter 4, Section 4.3.1, sub-section given in parentheses) and examples for using these impact model types for ecosystem services scenarios. The ecosystem services categories are adopted from Crossman et al. (2013).

| Impact model types of special relevance for IPBES | Examples of model applications to develop ecosystem services scenario | Main ecosystem services potentially addressed |
|---|---|---|
| Individual level models and evolutionary adaptation (4.3.1.1) | - | Provisioning<br>Cultural and amenity |
| Population models (4.3.1.2) | - | Provisioning<br>Cultural and amenity |
| Species distribution models/biogeography models (4.3.1.3.1) | Exploited marine species (Cheung et al., 2010), forestry revenues (Hanewinkel et al., 2013) | Provisioning<br>Cultural and amenity |
| Community-level models (4.3.1.4) | - | Provisioning<br>Cultural and amenity |
| Dynamic global vegetation models (4.3.1.6) | Carbon storage (Doherty et al. 2009, Friedlingstein et al. 2014) | Provisioning<br>Regulation<br>Habitat |
| Integrated assessment models (4.3.1.7) | Variety of ecosystem services, including food provision, water availability, carbon sequestration, flood protection (Stehfest et al., 2014) | Provisioning<br>Regulation<br>Habitat |

**Box 6.2:** Integrated assessment model (IAM) - the IMAGE 3.0 Framework

The IMAGE integrated assessment modelling framework provides an example of the use of the IAM approach in linking models for biodiversity and ecosystem services assessment. IMAGE was developed to understand how global, long-term environmental change and sustainability problems develop over time, driven by human activities such as economic development and population growth (Figure Box 6.2). Similarly to other IAMs, IMAGE can be used to identify problems of global environmental change, and to advise on possible response strategies. Earlier versions of the IMAGE model have been used to support various international assessments, including IPCC assessments, the United Nations Environmental Programme's (UNEP) Global Environment Outlooks, the Organisation for Economic Co-operation and Development's Environmental Outlooks and the Millennium Ecosystem Assessment (MA). Moreover, the model has been used extensively in the scientific literature.

In the IMAGE 3.0-GLOBIO Framework, models of socio-economic drivers, such as climate change, land-use change and pollution, are linked with models assessing impacts on the environment and biodiversity. The results of IMAGE-GLOBIO have provided information for policymakers at the international level on current biodiversity status and future trends (Alkemade et al., 2009). Specifically, IMAGE-GLOBIO has projected trends in biodiversity under future policy scenarios that involve multiple domains and drivers, including the expected outcome in the absence of additional policies to prevent biodiversity loss. IMAGE-GLOBIO delivers output in terms of Mean Species Abundance relative to the natural state of original species, land cover and land use (high resolution land use and land-use intensity based on GLC2000 and IMAGE), species richness index and wilderness area. Thus, IMAGE-GLOBIO allows the exploration of policy trade-offs between biodiversity conservation and the effectiveness of achieving goals in other domains.

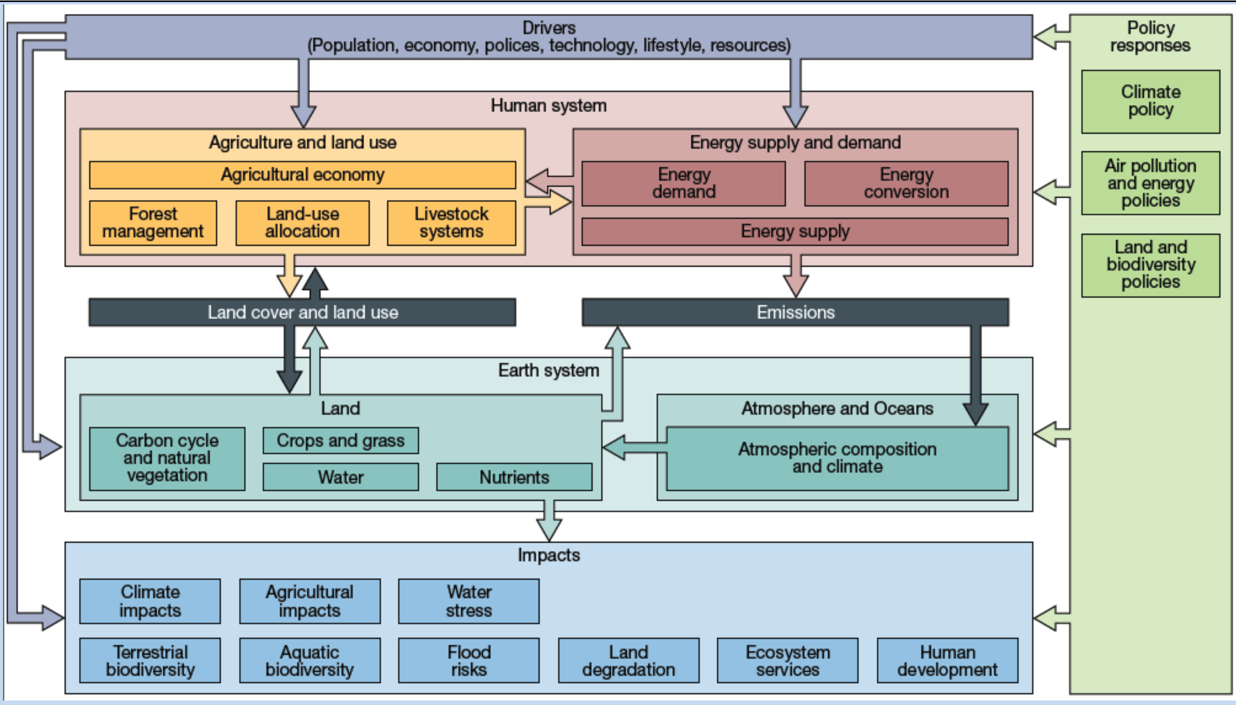



**Figure Box 6.2:** Framework of the IMAGE 3.0 Integrated Assessment Model (Modified from Stehfest et al., 2014).

Using the IAM framework to link models can address a broader range of questions related to biodiversity, such as trade-offs between climate change mitigation and biodiversity conservation. However, further work is needed to incorporate model components that represent more ecological processes, such as population dynamics or the biogeography of groups of animals (Harfoot et al., 2014).

Currently, the representation of biodiversity in IAMs is largely limited to terrestrial ecosystems (Chapter 4). The complexity of IAMs and the substantial resources needed to develop them may also render them less suitable compared with other, simpler methods of linking models.

## 6.3.2 Combining model and scenario outputs

Outputs from models and scenarios that are complementary in representing different domains and scales can be combined qualitatively so that each provides descriptions, projections or narratives of different axes of the biodiversity and ecosystem services assessment framework.

The projections or narratives generated by models and scenarios representing different domains can be combined to more holistically describe potential changes in social-ecological systems or a subset of the systems. Such linkages would be simple if the models or scenarios were coherent across scales and had the same analytical framework and logic. However, in many cases, models and scenarios may be constructed to be largely independent at different scales or domains but connected by the same issues they address or, conversely, they may be constructed at the same scales but address different issues. An iterative process is generally necessary to incorporate feedbacks and maintain storyline consistency, although feedbacks are seldom considered in this type of linkage. For the outputs to be compatible with one another, they should first be harmonised by categorising them under the same scenario archetype or family based on their drivers, assumptions, scenario logic and boundary conditions (Zurek and Henrichs, 2007; see Section 6.4).

# 6.4 Harmonising models and scenarios

## 6.4.1 Harmonising models across scales

Harmonising models to assess status and trends and project future changes in biodiversity and ecosystem services requires synthesising biophysical and socio-economic data and results that are available at different organisational, spatial and temporal scales and domains (Figure 6.2).

Scales can be defined considering two main properties: *grain* and *extent*. Grain refers to the resolution of the data, and extent to the size of the dataset. More specifically, the *organisational grain* is the resolution of the social, human or built capital information, the *spatial grain* is the size of the sampling unit, and the *temporal grain* is the data frequency. The extent refers to the size of the human system considered (*organisational extent*), the area (*spatial extent*) or period of time (*temporal extent*). For example, for an Enhanced Thematic Mapper plus sensor, the spatial grain is 30 meters (for bands 1 to 5), the temporal grain is 16 days (the satellite makes an image of the same place each 16 days), the spatial extent corresponds to a track 183 kilometres wide, and the temporal extent is the duration of the study (for example, a few days, a season or several years).

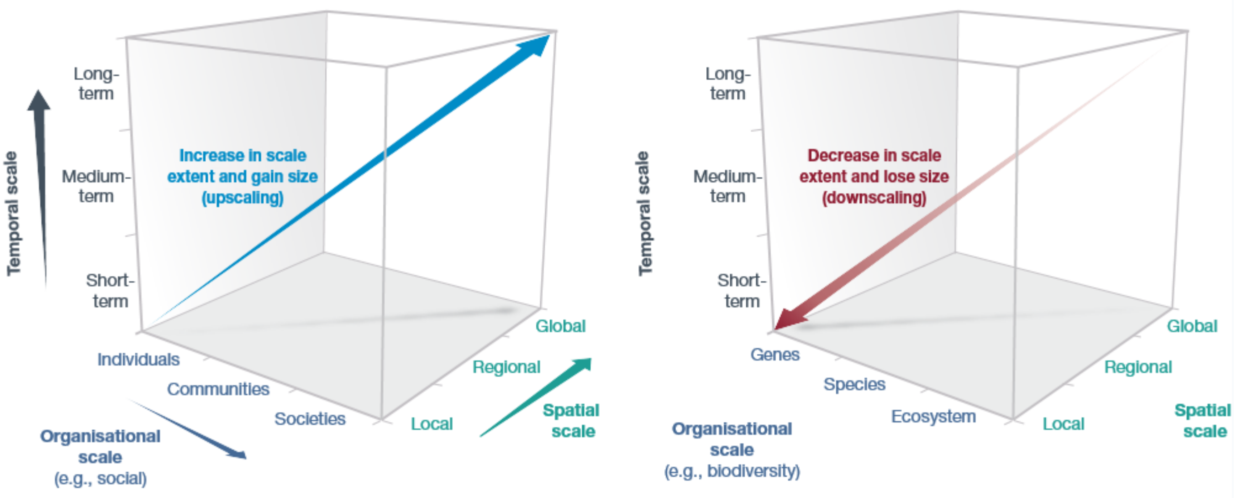



**Figure 6.2**: Spatial, temporal and organisational scales are usually correlated, thus the consequences of changing the scale of analysis (upscaling or downscaling) in any of these three dimensions need to be carefully considered. Upscaling (left panel) is related to an increase in scale extent and grain size, while downscaling (right panel) is the inverse process. Examples of organisational scale (or variables in Fig. 6.1) are: individuals, communities, societies for the social dimension; genes, species, ecosystems for biodiversity; and provisioning, regulating, cultural services for ecosystem services.

Space, time and organisational scales are usually correlated (Figure 6.2; Levin, 1992).

The assessment of large human systems or communities requires data at large spatial and temporal scales (but low resolution), and inversely data on specific local communities requires more localised temporal and spatial information, but at a high resolution. As a consequence of this spatial-temporal interaction, models with a coarse spatial resolution usually do not resolve processes that operate at fine temporal scales, and vice versa. For example, global-scale population dynamic models of fish do not resolve the fine-scale behavioural shift of individuals driven by changing local ecological or environmental conditions. There is an optimum scale for understanding specific natural dynamics of systems operating at a particular spatial scale (Wiens, 1989), and the challenge (for scientists and practitioners) is to identify the appropriate scale, for example by avoiding the introduction of too much detail into coarse-scale models.

Bringing model outputs to the appropriate scale is referred to as scaling, and can be done in two different directions: upscaling information from a local, fine-grained resolution to a global, coarse-grained resolution or, vice versa, downscaling the information.

Upscaling usually leads to an increase in the extent and decrease in the resolution, while downscaling increases the resolution of the data while losing the extent (Figure 6.3). Upscaling and downscaling are discussed in detail for organisational, spatial and temporal scales in Sections 6.4.1.1–6.4.1.3. In both directions, predictions are associated with errors and uncertainty, which are explored in the next section (Section 6.5).

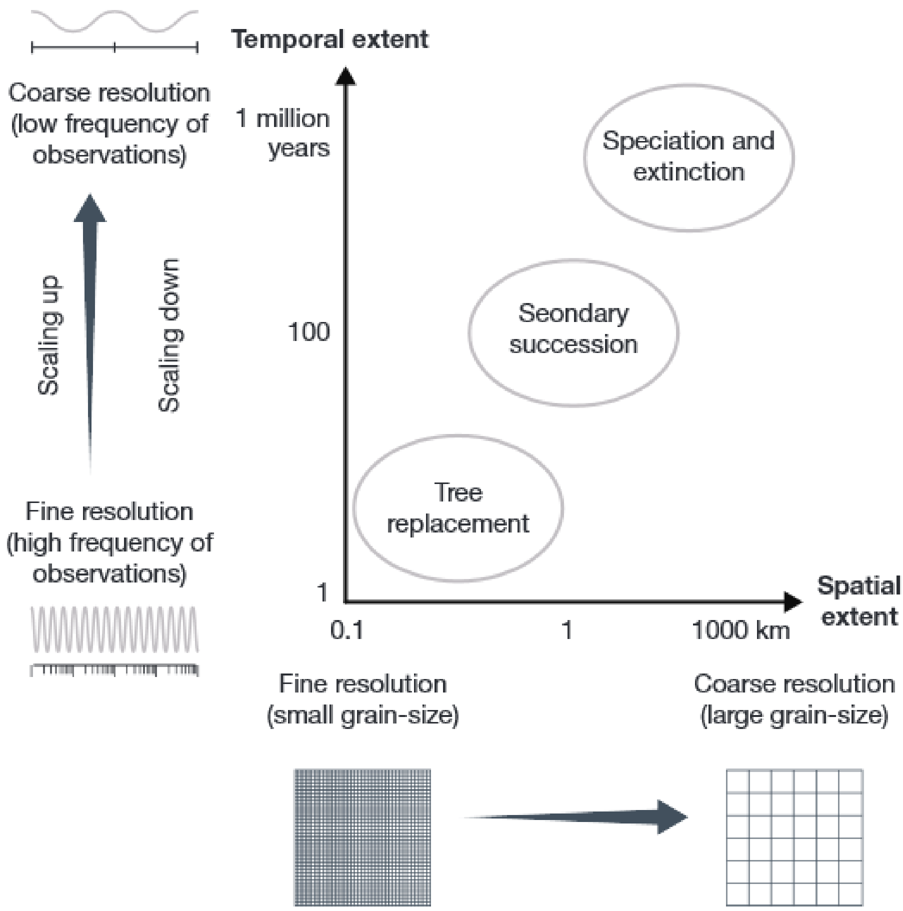



**Figure 6.3:** Four ecological processes (tree replacement, secondary succession, speciation and extinction) and their respective space-time domains. Large spatial and temporal extents are usually associated with coarse spatial and temporal resolution data, in contrast to small spatial and temporal extents. Consequently, scaling up is usually associated with an increase in spatial and temporal extent and decrease in resolution, while scaling down has as a consequence a reduction in extent, which should result in an increase in resolution. Note that these scales may be altered by direct and indirect drivers e.g., deforestation can reduce the rate of extinction.

Cross-scaling is appropriate in situations where biodiversity dynamics and ecological processes acting at a particular scale are also indirectly affected by processes acting at other scales (Section 6.4.1.4).

Local biodiversity patterns or ecosystem services, such as stocks and flows of water and other living resources, are mainly controlled by proximate factors acting locally, but are also affected by indirect global drivers of change (Levin, 1992), which would require data at a large spatial extent and for an extended period of time. Inversely, local actions affect the environment globally, and as a consequence the success of global scenario projections will depend on the congruence of scenarios and goals planned at more local scales (Cash et al., 2006).

#### 6.4.1.1 Social organisational scale

The underlying assumptions and mental models that people hold about the human-natural systems relationship drive the types of models and scenarios that are accepted and developed (Hamilton, 2011) (see also IPBES Deliverable 3d on 'diverse values and valuation', http://ipbes.net/).

Several models have been proposed and adopted to provide knowledge about human-natural systems in a range of spatial-temporal-organisational dimensions (Dietze et al., 2011). Some of these models are static with snapshot changes (e.g. computable general equilibrium), linear with projected changes over time (e.g. VISIT, Integrated Valuation of Environmental Services and Trade-offs (InVEST) (Goldstein et al., 2010; Kareiva et al., 2011; Dean et al., 2012) or system-based (e.g. World3 (Meadows et al., 1972), Global Unified Meta-model of the Biosphere (GUMBO) (Boumans et al., 2002) and Multi-scale Integrated Model of Ecosystem Services (MIMES) (Box 6.3) (Boumans and Costanza, 2007; Altman et al., 2014).

**Box 6.3:** Multi-scale Integrated Model for Ecosystem Services (MIMES) for the Manawatu watershed, New Zealand

The Manawatu River watershed is located on the North island of New Zealand and is home to about 200,000 people, with the land used intensively for agriculture, particularly dairy farming. Historically, the steep hills were forested, but the forest is now down to 20% of the original cover (Dymond, 2010).

In 2009, a newspaper article labelled the Manawatu the 'river of shame', as researchers had ranked it as the worst of 300 rivers tested for daily fluctuations in dissolved oxygen (Clapcott and Young, 2009). In response, the regional government initiated a collaborative process to bring together stakeholders, which became the Manawatu River Leadership Forum. This coincided in timing with the Ministry of Business, Innovation and Employment providing funding for Ecological Economics Research New Zealand to undertake the 'Integrated Freshwater Solutions' action research project.

A mediated modelling approach was used to support the collaborative effort to understand the underlying systems driving poor water quality (van den Belt, 2004), specifically those causing eutrophication, erosion and habitat destruction.

The mediated modelling scoping model was used to 'play out' some of the scenarios associated with the detailed 'Action Plan' signed off by the Manawatu River Leaders Forum. An example of one policy scenario is the provision of funding to reduce erosion by retiring land and planting trees as part of the Sustainable Land Use Initiative.

The mediated modelling effort *with* stakeholders was subsequently translated and enhanced to develop a spatially-explicit, dynamic MIMES (Altman et al., 2014). MIMES uses Simile software and links multiple databases in a way that allows the bundling and trade-offs of ecosystem services over time and space.

Here, erosion control (as undertaken for example by the Sustainable Land Use Initiative programme) is mapped to highlight the change in 'hotspots' over time and space (Crossman and Bryan, 2009). The progression of model development from mediated modelling to MIMES required a transition from interpreting stakeholder perceptions to more data-intense, specialist modelling by the science community.

Upscaling and downscaling along a social organisational scale requires an awareness of human-imposed boundaries and conventions, which often do not follow an ecosystem logic and may be difficult to clearly define (O'Brien and Vickerman, 2013).

For example, the use of surface water and groundwater can have a very different spatial extent. In addition, governing bodies are often guided by multiple ways in which their constitutions are divided in space; for example, the Auckland Council identified 30 different ways in which space is divided for water management (including water supply, water treatment, storm water, river/coastal and groundwater protection and various values from interest groups such as people from the Maori culture) (van den Belt et al., 2011). As such, optimum operational scales for models and scenarios differ in different organisational contexts and should be 'fit for purpose'.

An ecosystem services approach can be considered an organising principle for the harmonisation of models or their outputs along an organisational scale (Costanza et al., 1997; Daily, 1997; MA, 2005; Braat and de Groot, 2012).

An ecosystem services approach is also inherently multi-scale, as ecosystem services can be classified according to their spatial characteristics (Costanza, 2008; see Figure 6.4). (1) At a global level, climate regulation, carbon sequestration and storage as well as cultural or existence values do not depend on people's proximity to the ecosystems from where the services originate, whereas (2) local proximity is relevant for disturbance regulation/storm protection, waste treatment, pollination, biological control and habitat. (3) A directional flow characterises water regulation/flood protection, water supply, sediment retention/erosion control or nutrient regulation, (4) a point of use is relevant for soil formation, food/forest production and other raw materials, and finally (5) some ecosystem services and the benefits/values derived from them are related to the manner in which users move in space (and time), for example genetic resources, recreational potential and cultural values.

Due to their complexity, the scope for approaches aiming for optimisation is limited and the process of model building with stakeholders becomes as equally important as the model itself (van den Belt, 2004).

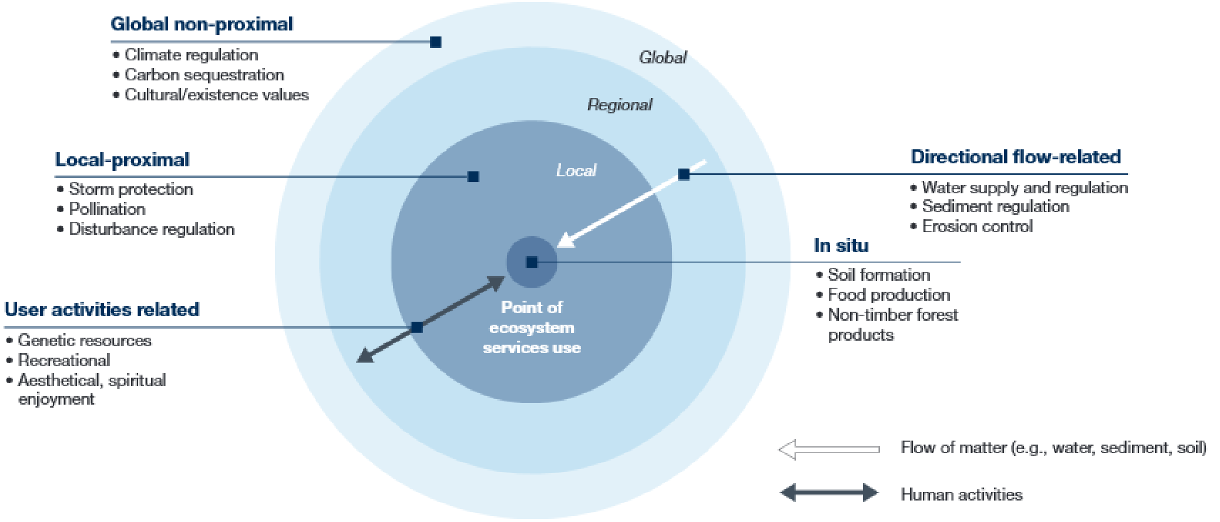


**Figure 6.4:** Multi-scale ecosystem services approaches classified according to their spatial characteristics (Modified from Biological Conservation, 141/2, Costanza, 2008, Ecosystem services: multiple classification systems are needed, 350-352, copyright 2008, with permission from Elsevier).

### 6.4.1.2 Spatial scale

*Downscaling*

Spatial downscaling is a common technique for providing spatial information for local conservation issues or management needs – such as establishing priority conservation areas – when high resolution information is not available (Rondinini et al., 2005; Bombi et al., 2012; Fernandes et al., 2014).

For example, downscaling is relevant to the incorporation of projections of climate models into local conservation planning (Wiens and Bachelet, 2010; Walz et al., 2014). There is a long history of developing downscaling methods for climate data that provides valuable experience for the downscaling of biodiversity and ecosystem services models and scenarios (Box 6.4). Downscaling has also been applied to biodiversity assessment, such as the downscaling of the RLI from the global to the national scale (Han et al., 2014; Rodrigues et al., 2014).

**Box 6.4:** Interpolation of local information with extracted global information

The introduction of global information by establishing statistical transfer functions is an efficient approach to improving the estimation error of ecological variables for locations where primary data are not available. A statistical analysis of precipitation data from 661 meteorological stations in China (Figure Box 6.4a) demonstrates that precipitation has a close relationship with topographic aspect, latitude, longitude and elevation. The statistical transfer function of mean annual precipitation under a Box-Cox transformation was derived as a combination of minimised residuals output by a method for high accuracy surface modelling with a geographically weighted regression using latitude, longitude, elevation, impact coefficient of aspect and sky view factor as independent variables. The introduction of spatial non-stationarity analyses into the interpolation of meteorological station data has greatly improved the interpolated climate surfaces (Figure Box 6.4c). For instance, inverse distance weighting was applied to the interpolation of mean annual precipitation in China for the 1960–2010 period, taking a digital elevation model (DEM) as secondary data (Figure Box 6.4b); the mean absolute error of the mean annual precipitation was 102.23 mm. The mean relative error of the interpolated mean annual precipitation decreased by 3% due to the combination of geographically weighted regression with inverse distance weighting; in addition, when high accuracy surface modelling was used, which displays a much better performance compared with classical methods such as inverse distance weighting, kriging and splines, the accuracy of the interpolated mean annual precipitation increased by 3% (Yue et al., 2013; Zhao and Yue, 2014). In other words, the introduction of both geographically weighted regression and high accuracy surface modelling can increase the accuracy of the interpolated mean annual precipitation by 6%.

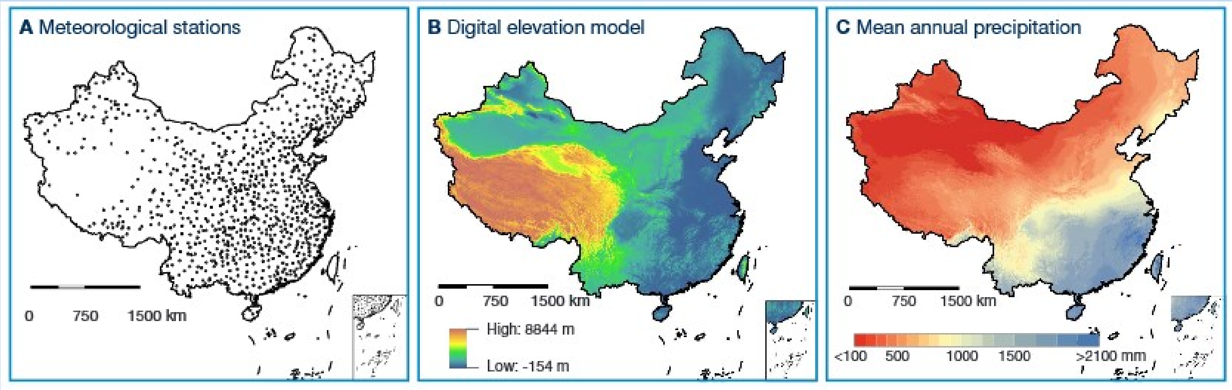



**Figure Box 6.4:** a) Spatial distribution of the meteorological stations with location information in China, b) digital elevation model of China, c) surface of mean annual precipitation in China, whereby the necessary global information is extracted from local information using geostatistics.

Most often, downscaling techniques are based on the interpolation of statistical relationships between specific model or scenario metrics and predictors with higher resolution data.

For example, the expected distribution of a species inside its geographic range can be inferred from high resolution data on the distribution of its habitat; thus downscaling the information to the scale of the habitat maps (e.g. Rondinini et al., 2005, Rondinini et al., 2011). A similar approach is obtained with hierarchical modelling (Keil and Jetz, 2014, Keil et al., 2013), which projects the relationship between coarse-grain species and environmental data onto a finer grain using fine-grain environmental (predictor) variables. This method was used for downscaling exploited fish and invertebrate distributions in Western Australia (Cheung et al., 2012). Similarly, Barwell et al. (2014) downscaled coarse-grained (> 100 km$^2$) Odonata atlas data to a more fine-grained (25km$^2$, 4km$^2$ and 1 km$^2$) local scale in mainland Britain, suggesting reasonable estimates of fine-grain occupancy, with varying errors according to species traits. Recent studies have shown the high predictive performance of downscaling models compared with field observations of invasive alien species (Fernandes et al., 2014), birds (Keil et al., 2013), Sardinian reptiles (Bombi et al., 2012), and global marine circulation (Sandø et al., 2014). Specific methods such as the hierarchical Bayesian modelling (HBM) approach are shown to improve the performance of downscaling compared with other statistical methods (Keil et al., 2013). These predictions may be further improved when combined with macro-ecological relationships (e.g. scale-area relationships) (Keil et al., 2013).

Dynamic downscaling that is based on mechanistic models may be more appropriate than statistical downscaling in systems where the relationship between coarse-scale and fine-scale dynamics are complex and non-linear or observational data are insufficient.

Dynamic downscaling uses fine-resolution dynamic models to estimate fine-scale dynamic features (Stock et al., 2011). Available downscaling methods involve developing fine-scale models that are forced with coarse global simulations, or forcing a fine-resolution model component with information from a coarse resolution model. The coupling between coarse-scale and fine-scale models can be 'one-way' or 'two-way'. Dynamic downscaling has been applied widely in regional climate and oceanographic modelling. On the one hand, dynamic downscaling offers consistency and reliance on the fundamental principles of physics, chemistry, biology and ecology. On the other hand, it requires a higher computational cost to run the models. Also, the coupled coarse/fine-scale models are more complex and costly to develop. Furthermore, while dynamic downscaling may improve the representation of fine-scale dynamics, it is still strongly influenced by any bias in the coarse-scale simulations used for the boundary forcing.

While the downscaling methodology is well developed, there is a trade-off between the cost of collecting data and developing models at a fine scale on the one hand, and the uncertainty of downscaled outputs on the other.

The decision of which scale to adopt ultimately depends on the resources available and the acceptable level of error, which can be quantified by validating the downscaled model with sampled high-resolution data.

*Upscaling*

Environmental consequences of human activities sometimes encompass broad spatial and temporal scales, which need global assessments and policy actions. For this reason, it is often necessary to transfer local high-resolution data to broader scales, which is called spatial upscaling (Flint and Flint, 2012).

Upscaling methods are more intuitive than downscaling ones, as they involve extrapolating values over a larger space. Upscaling methods can be categorised into four main types: coarse graining, lagging, accumulating and rating (Figure 6.5). Coarse graining increases the spatial extent of the unit, lagging increases the separation between units, accumulating sums all the finer scale values within a larger spatial extent, and rating compares data on distinct units that differ in terms of size or some other characteristic to develop scaling functions (Schneider, 2009).

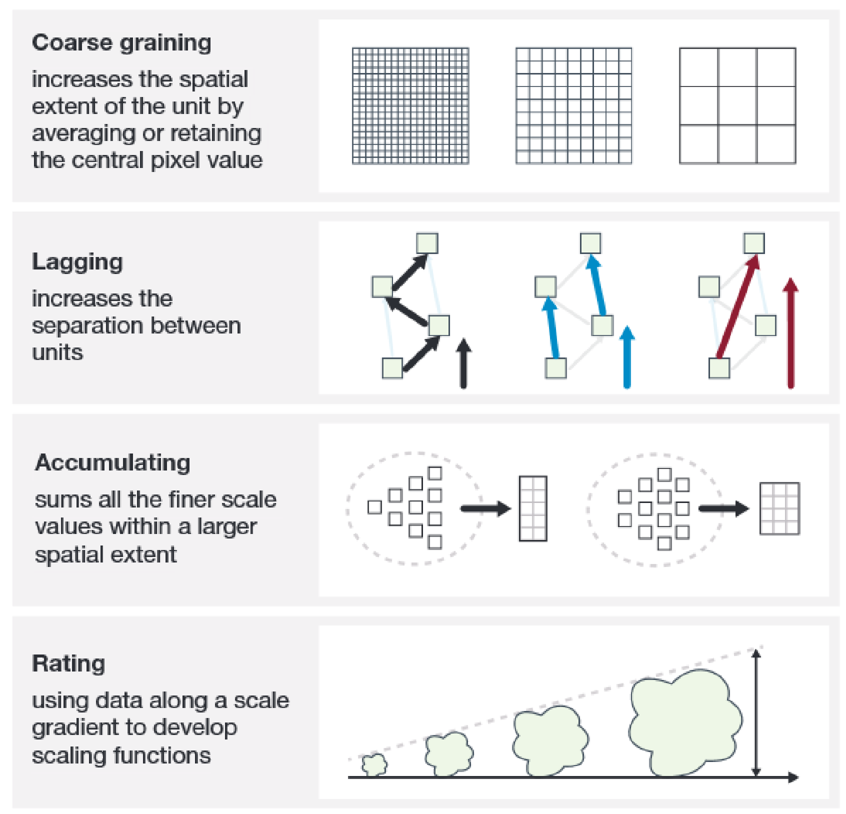



**Figure 6.5:** Comparison of the four main upscaling methods (Modified from Academic Press, Schneider, 2009, Quantitative ecology: measurement, models and scaling, copyright 2009, with permission from Elsevier).

Spatial upscaling approaches have been commonly applied to analyse satellite imagery and to combine statistical and image processing analyses with simulation models and field observations (Zhang et al., 2007; Chen et al., 2010; Fu et al., 2014).

For example, upscaling has been used to estimate net ecosystem exchange or carbon dioxide fluxes from flux towers at the landscape and regional scales (Fu et al., 2014). In another example, a simple exponential relationship between Leaf Area Index (LAI) and the Normalised Difference Vegetation Index (NDVI) obtained with a Landsat image was used to upscale LAI values to Arctic landscapes (Williams et al., 2008). Other methods have been developed to upscale gross ecosystem production (GEP) from leaf or stand levels to larger regions (ca. 12 $km^2$) taking into account tree canopy structure (Hilker et al., 2008), using Light Detection and Ranging (LiDAR) images. Results showed a high correlation ($r^2$ between 0.75 and 0.91, $p < 0.05$) between estimated and measured ecosystem production. A good fit between upscaled estimated values and field measurements was also obtained with net primary productivity in China, showing that the integration of field data with remote sensing through an ecosystem model can generate reliable estimates (Zhang et al., 2007). Upscaling can result in better estimates than those obtained from coarse-grained resolution images, such as obtained from satellite remote sensing data from the Moderate Resolution Imaging Spectroradiometer (MODIS) (Fu et al., 2014), possibly because it can integrate the variability observed at finer scales in the coarse-scale evaluation. Similar results were obtained by Hay et al. (1997) when upscaling forest stand characteristics with image resampling techniques. This showed that appropriately upscaled satellite imagery can represent a more accurate estimation than an image obtained at the upscaled resolution. However, it is costly to obtain fine-scale information over a large extent for upscaling to reveal a broad scale pattern.

In the process of upscaling, quantitative approaches to preserve the quality of the original information should be applied whenever possible; otherwise it can contribute to scaling uncertainties (Section 6.5).

For example, using an approach called modelled net ecosystem exchange, the mean, variance and skewness properties of the fine-scale NDVI in an Arctic tundra landscape are preserved (Stoy et al., 2009).

#### 6.4.1.3 Temporal scale

Quantifying and forecasting temporal changes in biodiversity and ecosystem services by linking and harmonising models and scenarios at an appropriate scale is important not only to address ecological issues, but also to develop policies and achieve global conservation goals.

The appropriate temporal scales for models and scenarios vary (daily, monthly, annual, decadal and centennial), depending on the properties of the direct and indirect drivers (Chapters 3 and 4), the mechanisms through which these drivers result in changes in biodiversity and ecosystem services, and the policy context.

*Upscaling*

Upscaling methods across temporal scales are in principle similar to those for spatial scales, and include accumulating, coarse-graining, lagging and rating (Figure 6.5).

Accumulating involves summing data or model outputs from finer temporal intervals to present longer-term average conditions. For example, species occurrence records are accumulated over a long time period (e.g. multiple decades) before being used to predict a current distribution range. Coarse-graining involves averaging estimates over smaller temporal units, for example averaging annual climate data to calculate climatology. For lagging, multiple snapshots are used to present changes over a longer time period. In some cases, these snapshots could be generated from a diversity of sources, including formal quantitative measurements, model outputs or expert knowledge. For rating, quantitative functions are developed to rescale finer temporal resolution data to estimates over a longer time period. For example, time trends of point information on soil solution data have been scaled by linking them to soil chemical data which was available at a higher temporal resolution, using both statistical and process-oriented methods (Zirlewagen and von Wilpert, 2010).

When fine-scale data or outputs are available, the cost of temporal upscaling is relatively low; however, the temporal characteristics of the data across scales should be considered carefully.

Otherwise, upscaling may contribute to scaling errors (see Section 6.5). Specifically, the temporal variance of the data may be smoothed after upscaling. For example, seasonal differences in net primary production will not be represented in annual averages. On the other hand, higher internal variability may also need larger temporal samples for upscaling.

*Downscaling*

The downscaling of temporal data is primarily based on numerical (mechanistic) models, statistical analysis and stochastic algorithms.

For instance, Rebora et al. (2006) developed a new spatial-temporal downscaling procedure for flood forecasting, called RainFARM, as an alternative to stochastic algorithms. RainFARM generates small-scale rain rate fluctuations that preserves the spatio-temporal evolution of rainfall patterns. Mendes and Marengo (2010) proposed an alternative to numerical models, developing a temporal neural network for downscaling global climate outputs (downscaling daily precipitation time series). A novel conceptual and analytical model of biodiversity loss based on the landscape ontogeny, Terragen, is currently being developed by Rosa et al. (2013) and aims to generate biodiversity scenarios for the humid tropics, partially based on the downscaling of temporal data on biodiversity loss and deforestation models. However, most existing examples of temporal downscaling are related to the modelling of drivers such as climate, while examples of biodiversity and ecosystem services models and scenarios are limited.

#### 6.4.1.4 Cross-scale interactions

When system processes (biophysical and/or social-economic) interact across scales (spatial, temporal or organisational), resulting in non-linear dynamics, the harmonisation of models and their outputs across these scales is more likely to result in scaling errors (Peters et al., 2007), see Section 6.5.2). In such cases, the use of multiple-scale models performs better than single-scale models.

For example, Boscolo and Metzger (2009) showed that multi-scale models that consider pattern-process relationships at different extents in a unique model always perform better than single-scale models in predicting the occurrence of bird species in a tropical forest. This is probably because extinction and recolonisation processes that control species occurrence act simultaneously at different scales. There are many other examples of important ecological processes that are modulated by processes that interact across scales, such as bark beetle eruptions (Raffa et al., 2008), parasitism (Tompkins et al., 2011), fire disturbances (Falk et al., 2007), and runoff and erosion processes (Allen, 2007) (see Box 6.5).

**Box 6.5:** Regional assessments that include both global and local information

Ecosystem services are controlled by a combination of global and local factors. The system dynamics that generate the ecosystem services cannot be recovered from the global or local controls alone (Phillips, 2002). In this box, we illustrate how harmonising models across multiple scales can improve the accuracy of the model outputs.

Results from a satellite-observation-based approach (global scale) (Piao et al., 2009) and a local-information-based approach (local scale) (Fang et al., 2001) were combined by means of high accuracy surface modelling (Yue, 2011). China's national forest inventory database from 2004 to 2008 includes 160,000 permanent sample plots and 90,000 temporary sample plots scattered over the land surface of China. The cross-validation comprised four steps: 1) 5% of the sample plots of each forest type in each province were removed for validation prior to model creation; 2) the spatial distribution of average forest carbon stocks (CS) in China during the 2004–2008 period was simulated at a spatial resolution of 5km×5km using the remaining 95% of sample plots; 3) the mean absolute error and mean relative error were calculated using the 5% validation set; and 4) the 5% validation set was returned to the pool of available sample plots for the next iteration and another 5% validation set was removed. This process was repeated until all the sample plots had been used for validation at least once and the simulation error statistics for each sample plot could be calculated.

The mean absolute errors of the carbon stock surfaces generated by the satellite-observation-based approach (Figure Box 6.5a) and the Kriging (Figure Box 6.5b) were respectively 1.9 and 2.0 $kg{\cdot}m^{-2}$ respectively. When the local information was combined into satellite-observation-based approach by means of high accuracy surface modelling (Figure Box 6.5c), the mean absolute error was decreased to 0.9 $kg{\cdot}m^{-2}$. The mean relative errors of both the global and local-information-based methods were reduced by at least 53% because the local and global information was fused by means of high accuracy surface modelling.

Based on the high accuracy surface modelling, the annual mean CS of all forest types in China was 7.1 Pg during the 2004–2008 period, given contributions of 2.7, 4.0 and 0.4 Pg from coniferous, broadleaf and mixed forests respectively. Similarly, the annual mean carbon density was 4.6 $kg{\cdot}m^{-2}$ during the 2004–2008 period, with contributions of 4.4, 4.7 and 4.2 $kg{\cdot}m^{-2}$ from coniferous, broadleaf and mixed forests respectively. The satellite-observation-based approach underestimates annual mean CS, whereas Kriging overestimates the annual mean CS of China.

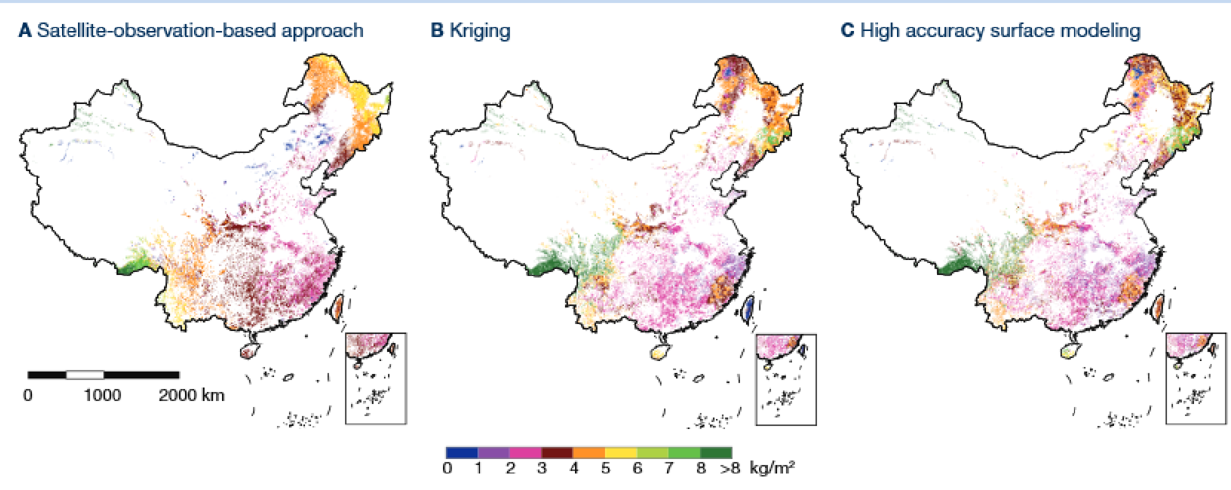



**Figure Box 6.5:** Surfaces of carbon stocks created by different methods: a) satellite-observation-based approach, b) Kriging, and c) high accuracy surface modelling.

Multi-scale models can also be used to represent interactions between human organisational scales in the assessment of ecosystem services.

The management of salmon resources in the Columbia River Basin, USA, is a good example (Rieman et al., 2001). Conflict within and among groups of individuals and organisations that have different interests, values and power can be viewed as an interacting hierarchical structure. In particular, the interests of local loggers, fishers and environmentalists conflict with the interests of those planning hydropower utilities, as well as pitting native fishers against offshore fishermen and environmental groups (Rieman et al., 2001), resulting in non-linear dynamics in the social-ecological system (Peters et al., 2007).

The use of multi-scale modelling approaches can help to consider the different factors and their interactions that are important at different scales. A drawback to using multi-scale modelling is that cross-scale interacting processes are difficult to model accurately and may result in error propagation as model complexity increases (see Section 6.5.2).

### 6.4.2 Harmonising scenarios

To compare, synthesise or combine existing assessments of biodiversity and ecosystem services that use scenarios with different objectives, policy questions, assumptions, uncertainties, or focus on different temporal and spatial scales, the scenarios first need to be harmonised.

Scenarios that are related to biodiversity and ecosystem services have been produced and used in different international (e.g. MA), national (e.g. the United Kingdom's Alternative Future Scenarios for Marine Ecosystems, Pinnegar et al. (2006) and local assessments (e.g. Manawatu basin management, Box 6.3). The differences in objectives (for example to assess greenhouse gas emissions or sustainability in fisheries) and assumptions, and the use of different scenario development methodologies may render direct comparison between these scenarios difficult. Many initiatives employ different methodologies in developing scenarios, even in different iterations of the assessment, depending primarily on the goals, spatial scales, social-economic and policy context, and the resources available for the scenario development exercises (Biggs et al., 2007) (see Chapter 3). However, these scenarios may need to be combined for comprehensive assessments that include different elements and domains relating to biodiversity, ecosystem services and human well-being (Figure 6.1). Harmonisation of these scenarios thus becomes important.

Existing scenarios belonging to the same archetype or family can be harmonised to provide more comprehensive descriptions of possible futures (Biggs et al., 2007).

Available literature on standardising and harmonising scenarios for environmental assessments suggests three main steps: 1) identify and discuss the application of the scenarios and their main characteristics, 2) compare the key assumptions and storylines behind the scenarios, and 3) compare the trends observed in the main scenario methodology in relation to policymaking (Van Vuuren et al., 2012). Scenarios can be categorised into 'scenario families' or archetypes according to their underlying assumptions, storyline, logic and characteristics. Some of the key assumptions and variables in which these scenarios differ include risk-perception of and resulting policy actions in response to environmental change, the spatial scale of drivers and systems and their trends, and the degree of cooperation in the society (Biggs et al., 2007; Van Vuuren et al., 2012).

Scenarios describing plausible futures for different spatial scales can be harmonised, although the existing literature largely discusses methods for downscaling. To downscale scenarios (Biggs et al., 2007), scenario pathways at a large scale can be used as boundary conditions to frame developments in finer-scale scenarios.

This ensures that the outcomes of the regional scenarios do not conflict with those of the global scenarios. Also, in some cases, and with the help of expert and/or stakeholder participation, large-scale scenario pathways can be contextualised and applied to specific regions or issues. For example, different Shared Socio-economic Pathways (SSPs) developed for the IPCC were converted into different scenario pathways for oceans and fisheries through an interdisciplinary expert workshop. Moreover, scenarios at different scales may be developed without much reference to one another but can then be mapped together (see Section 6.4). In other cases, large-scale scenarios can be applied directly to examine regional policies without the need to develop complete regional scenarios, for example the application of Representative Concentration Pathways (RCPs) to assess climate change impacts at both global and regional scales.

The mapping of scenarios onto archetypes or families could be facilitated using tabular or graphical representation.

For example, existing scenarios for global environmental assessments include the Global Scenario Group (GSG)'s work on great transitions (Raskin et al., 2002, Raskin, 2005), the IPCC Special Report on Emissions Scenarios (SRES) (Nakićenoić and Swart, 2000), UNEP's Third Global Environmental Outlook (GEO3) (UNEP, 2002) and the World Water Vision work (Cosgrove and Rijsberman, 2014; Van Vuuren et al., 2012) (Table 6.3). To harmonise these scenarios, they are first characterised by eight broad attributes: economic development, population growth, technological development, main objectives, environmental protection, trade, policies and institutions, and vulnerability to climate change (rows in Table 6.3). Based on these attributes, they can be categorised into different archetypes or scenario families: global sustainable development, business-as-usual, regional competition, economic optimism, reformed markets and regional sustainability (columns in Table 6.3). The IPCC has developed multiple sets of socio-economic scenarios for different assessment reports, such as SRES (developed in the Fourth Assessment Report) and SSPs (developed in the Fifth Assessment Report) (O'Neill et al., 2014). These IPCC scenarios can be characterised and mapped graphically according to the underlying socio-economic challenges for mitigation and adaption under each scenario (Figure 6.6).

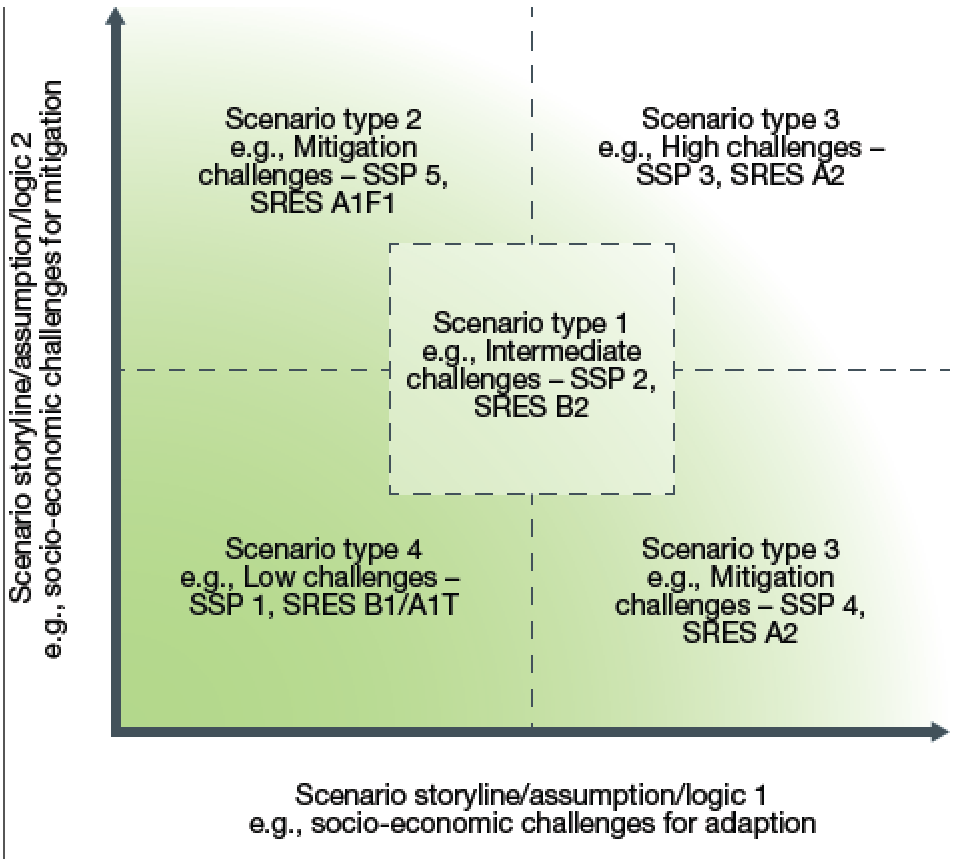


**Figure 6.6:** An example illustrating the mapping of scenarios onto scenario families or archetypes based on the storyline, assumption and logic of the scenarios. The example concerns the mapping of the Special Report on Emissions Scenarios (SRES) and Shared Socio-economic Pathways (SSPs) developed by the IPCC (Modified from Global Environmental Change, 22/4, Kriegler et al., 2012, The need for and use of socio-economic scenarios for climate change analysis: A new approach based on shared socio-economic pathways, 807–822, copyright 2012, with permission from Elsevier).

**Table 6.3:** Archetypes or families of scenarios from previous global environmental assessments and their key characteristics and assumptions (Modified from Global Environmental Change, 22/4, Van Vuuren et al., 2012, Scenarios in Global Environmental Assessments: Key characteristics and lessons for future use, 884-895, copyright 2012, with permission from Elsevier).

| Archetype/ scenario family | Global sustainable development | Business as usual | Regional competition | Economic optimism | Reformed markets | Regional sustainability |
|---|---|---|---|---|---|---|
| Economic development | Ranging from slow to rapid | Medium | Slow | Very rapid | Rapid | Medium |
| Population growth | Medium | High | Low | Low | Low | Medium |
| Technological development | Ranging from medium to rapid | Medium | Slow | Rapid | Rapid | Ranging from slow to rapid |
| Main objectives | Global sustainability | Not defined | Security | Economic growth | Various goals | Local sustainability |
| Environmental protection | Proactive | Both reactive and proactive | Reactive | Reactive | Both reactive and proactive | Proactive |
| Trade | Globalization | Weak globalization | Trade barriers | Globalization | Globalization | Trade barriers |
| Policies and institutions | Strong global governance | Mixed | Strong national governments | Policies create open markets | Policies target market failures | Local action |
| Vulnerability to climate change | Low | Medium | Mixed – varies regionally | Local action | Low | Low |
| **Examples** | | | | | | |
| SSP | SSP1 | SSP2 | SSP3/SSP4 | SSP5 | | |
| SRES | B1 (A1T) | B2 | A2 | A1F1 | | B2 |
| GEO3/GEO4 | Sustainability first | | Security first | Market first | Policy first | |
| Global scenario group | New sustainability paradigm | | Barbarization | Conventional world | Policy reform | Eco-communalism |
| Millennium Ecosystem Assessment | Techno-garden | | Order from strength | | Global orchestration | Adapting mosaic |

Some of the methods for downscaling scenarios can also be applied to upscale scenarios from finer to broader spatial scales. However, existing examples of scenario scaling have a greater emphasis on downscaling than on upscaling, limiting the available experience that could be drawn on.

Generally, to upscale finer scale scenarios to a large spatial scale, teams of developers can collaboratively develop finer scale scenarios that are consistent across regions. These local and regional scale scenarios then collectively provide a description of the future at the global level. This method of upscaling can minimise conflict between the local scale context and larger scale assumptions, while continuing to represent the diversity of the local scale context. However, substantial resources and effort are needed to coordinate the development and aggregation of multiple local-scale scenarios. In the case of multi-scale scenarios, different scenario components are kept at their most appropriate scale (space and time) with linkages between scales being established upfront (Biggs et al., 2007).

## 6.4.3 Model benchmarking

Benchmarking is the process of systematically comparing sets of model predictions with measured data to evaluate model performance. It should also help identify processes that may be poorly represented in models (McCarthy et al., 2012).

Benchmarking is common practice in fields other than ecology: for example, global circulation models included in IPCC reports need to be able to hindcast historical temperature trends derived from multiple sources. In scenario work, model predictions are sometimes weighted by the model performance in relation to the benchmarks (e.g. Rammig et al., 2010). General guidelines for benchmarking environmental models have been developed by Bennett et al. (2013) and a particular framework for land (ecosystem) models by Luo et al. (2012), but we are not aware of any multi-model benchmarking activity with species-level biodiversity models or ecosystem services models. The framework proposed by Luo and colleagues as part of the International Land Model Benchmarking (ILAMB) project includes 1) an evaluation of targeted aspects of model performance, 2) a set of benchmarks as defined references to test model performance, 3) metrics to measure and compare performance skills among models, and 4) model improvement. To improve the credibility of species-based biodiversity and ecosystem services models, benchmarking should be further developed. Species distribution models, for example, could be tested against observed historical changes in species ranges (Chen et al., 2011). This also highlights the need for good empirical data such as from remote sensing (e.g. satellite products and aerial photos).

Benchmarking should be accompanied by standardised model documentation and the archiving of model source codes, input data, model results and model result processing tools.

For biogeochemical models, such as global terrestrial carbon cycle models, guidelines for developing standardised archives were suggested by Thornton et al. (2005). For biodiversity and ecosystem services models, the current situation is unsatisfactory. Even though the results from numerical models should in principle be 100% reproducible, this is often not the case, for example because complex models are often under constant development, implying that references to published model descriptions are outdated. Archives for this purpose still have to be developed (Thornton et al., 2005).

## 6.5 Uncertainty in linking and harmonising models

### 6.5.1 Cascade of uncertainty from models linking biodiversity and ecosystem services

Uncertainties originating from different biodiversity and ecosystem services models that are linked across spatial and temporal scales, elements and domains may potentially propagate through the chains of models, affecting the ultimate envelope of uncertainty (Figure 6.7).

To allow the application of model linkages to provide useful outputs to inform and assist decision making on biodiversity and ecosystem services issues, sound estimates or assessments of uncertainty are needed (Dunford et al., 2014). The typology of uncertainties is described in Chapter 1, while details of uncertainties associated with specific model components are described in Chapters 2–5.

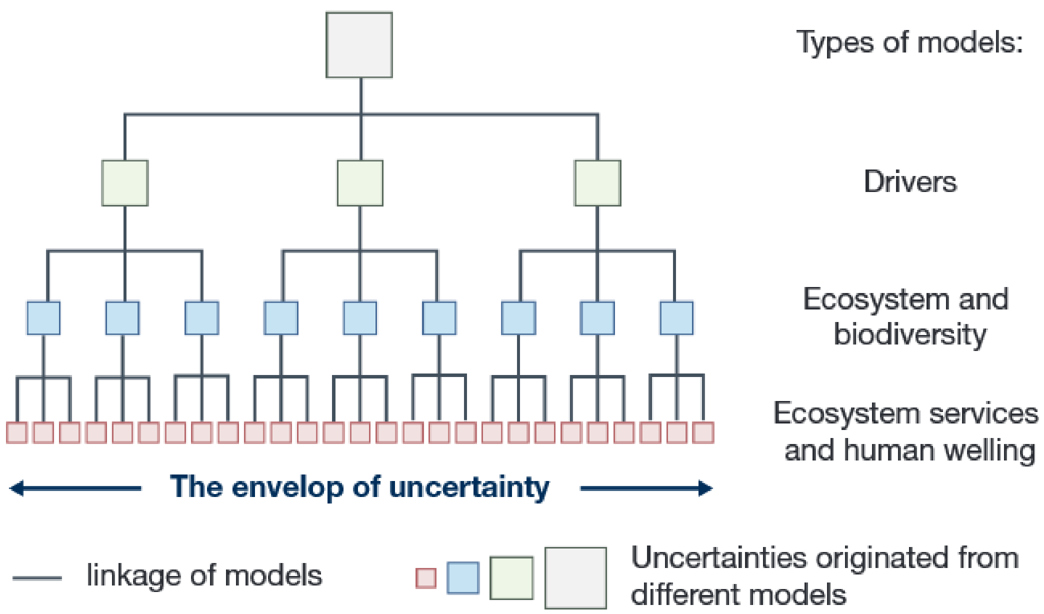


**Figure 6.7:** Cascade of uncertainties linking drivers, biodiversity and ecosystem services and human wellbeing models.

The width of the envelope of uncertainty depends on the nature of the interactions between linked models and their uncertainties.

The types of interactions include linearity of the linkages, the existence of threshold responses and positive/negative feedbacks (Peters and Herrick, 2004). When the processes linking two or more models are non-linear, uncertainties may be dampened or magnified through model linkages, for example through attenuation or amplification of changes in higher trophic level production in marine ecosystems driven by climate change (Chust et al., 2014; Stock et al., 2014). In a special case of non-linearity in which thresholds in triggering responses between models exist, the envelope of uncertainty may become more difficult to explore as thresholds are often difficult to specify. Feedbacks in social-ecological systems can be positive or negative, and uncertainties propagated in models that are linked dynamically with feedbacks result in emergent dynamics that are difficult to predict. Ignoring or mis-specifying the types of linkages will reduce the reliability of the linked model outputs.

Available options to address errors associated with linking models include not linking the models, limiting the extent of model linkages, and exploring the envelope of uncertainty resulting from model linkages (Peters and Herrick, 2004).

Selection of these options requires careful consideration of the necessity and marginal benefits of model linkages and the trade-offs in errors between model over-simplification and the increased uncertainty from more complex models; this requires the systematic exploration of different types of uncertainties associated with the linked models. Such exploration involves formal numerical approaches (e.g. comparison of model outputs with past observations and/or analysis of large model ensembles) and/or expert judgment (Dunford et al., 2014). For example, in a cross-sectoral, regional-scale Integrated Assessment Platform for the assessment of climate change impacts, the use of well-designed approaches to combine numerical analysis and expert opinion in addressing model uncertainties could improve the usefulness of the model outputs for decision making and the understanding of the uncertainty associated with it (Dunford et al., 2014).

The limited availability of observational data sets may make it difficult to evaluate the reliability of outputs from linked models. In particular, data is challenged by issues of consistency between temporal and spatial scales and confounding effects of multiple human pressures such as climate change and fishing.

The limitations of available data should not prevent application of the models nor deem all model projections unreliable, as projections also gain credibility through their reliance on robust ecological and physiological principles. It should, however, temper interpretation of the results.

## 6.5.2 Scaling errors and uncertainty

In downscaling or upscaling observations, models or their outputs, the wider the order of magnitude of scale being harmonised, the higher the risk of propagating errors (Jarvis, 1995).

A change in spatial or temporal scale results in a change in heterogeneity in the patterns, with heterogeneity increasing with a finer grain (given a constant extent) or a larger extent (given a constant grain). Scaling in systems with a gradual transition in grain and extent, or in systems that are scale-invariant (such as fractal systems), is usually simple and can be done using relatively simple regression functions. For example, it is well known that the size and frequency of disturbances are inversely related (e.g. large-scale disturbances are less frequent than small-scale disturbances), and this can be easily represented by a power law function (White et al., 2008). However, scaling between two or more scale levels, where non-linear changes in heterogeneity occur, may be much more challenging to apprehend using simple mathematical models, and thus lead to significant error propagation. Thus, harmonisation across a wide range of scales or with large heterogeneity in grain and/or extent is not recommended (Wiens, 1989). National or local policymakers should be cautious in relying on downscaled data to made decisions that are sensitive to high scaling errors, such as spatial planning (see Chapter 2).

Upscaling or downscaling models with processes that interact between different spatial or temporal scales will increase the scaling error.

The carbon flux from woody debris, for example, is simultaneously affected by climate, site environment and species-specific variations in wood characteristics (Weedon et al., 2009), as well as by the interactions of those processes that occur at different spatial and temporal scales. As a consequence, any upscaling or downscaling framework will need to consider interactions between these processes to properly model carbon dynamics. It could therefore be useful to consider species traits that regulate wood and decomposition characteristics at a more local (plot) scale, even in global terrestrial carbon cycle models (Weedon et al., 2009).

As a result, it is important to understand and identify any thresholds of scaling above which fundamental shifts in underlying processes that regulate the studied system occur (Wu et al., 2006). In these cases, it may be necessary to invest in mechanistic scaling approaches that explicitly model and represent the interactions between scales in the system.

An additional source of scaling error is the incorrect use of scaling functions, particularly in predicting species distributions. Such errors can be reduced through the careful selection of modelling and scaling methods.

For example, spatial aggregations of organisms can lead to bias in estimates of their abundance if the scaling process is non-linear (Stoy et al., 2009). Errors are commonly more severe when projecting the absence and occurrence of the organisms compared with their global range. Indeed, downscaling usually tends to lead to an overestimation of species distributions (Sardà-Palomera et al., 2012). However, precise information on species distributions at the local level is crucial for local decision making (Franklin et al., 2013), for example to identify biodiversity hotspots (Sardà-Palomera et al., 2012). In these cases, a more complex framework that combines niche and spatial models with spatially-explicit fine-grain approaches is necessary to reduce errors when modelling species locations (Azaele et al., 2012). Different techniques have been proposed to deal with species' spatial aggregation, such as the scale transition theory (Melbourne and Chesson, 2006) and the shot noise Cox processes, which allow a better prediction of population estimates at fine scales starting from coarser ones ((Azaele et al., 2012).

Ground observations and global models at coarse spatial resolutions are important sources of data for simulating changes in biodiversity and ecosystem services (Box 6.4). However, too sparsely distributed ground observations are often unable to satisfy the data requirements of regional or local stakeholders and decision makers. One major problem concerns how to estimate values for locations where reliable estimates cannot be generated by interpolation.

Many global models are difficult to use at regional and local levels because their spatial resolutions are too coarse or because important region-specific processes are missing.

For regional applications, regionally-tested downscaled global models or region-specific and site-specific models have to be developed (e.g. Hickler et al., 2012; Seiler et al., 2014). High-quality ground observation data and model benchmarking at the desired scale are crucial for both of these approaches.

## 6.6 Conclusions

Because of the complexity of the systems relevant to assessing the current status of and trends in biodiversity and ecosystem services and for developing future scenarios, it is often necessary to link models or scenarios representing their different components. Models and scenarios that integrate feedbacks and trade-offs across temporal and spatial scales and among dynamic societal economic and natural systems can address particularly complex challenges and guide decision making. Ultimately, the question of whether, how and to what extent biodiversity and ecosystem services models should be harmonised and linked depends on the research and policy objectives. The integration of quantitative models, qualitative approaches and expert knowledge has great potential to advance our understanding and predictions of biodiversity and ecosystem services. To facilitate the development of methods for linking and harmonising scenarios and models, we need to build communities of multi-disciplinary researchers and practitioners to support such research and decision support.

The rapidly growing number of model intercomparison projects provides an opportunity for fostering the harmonisation of models and cultivating a community to make advancements in the long term. However, existing intercomparison projects have a sectoral focus, for example on carbon cycling, forest productivity, agriculture or fisheries. Strengthening the linkages between biophysical and human domains is a major challenge. Although increasing efforts are being made in this area, such as IAMs and ecosystem services assessments, the more extensive development and application of these approaches should be encouraged to accelerate the state-of-the-art in linking models and scenarios across social and natural domains.

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# 7 Building capacity for developing, interpreting and using scenarios and models

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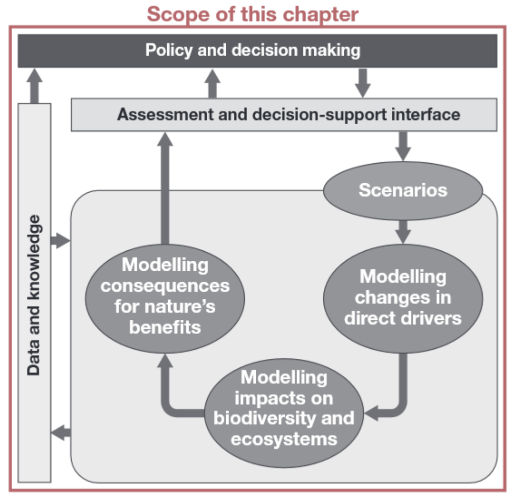
**Purpose of this chapter:** Critically reviews key challenges and potential solutions for building capacity in the development and use of scenarios and models (covered in Chapters 2 to 6) across different scales and regions and across a wide range of policy and decision-making contexts. This chapter also provides guidance on strategies to develop capacity for effective participation in the development and use of scenarios and models in IPBES assessments.

**Target audience:** A broader, less technical audience than for many of the other chapters in this report, but aimed particularly at readers seeking guidance on how best to build capacity in developing and using scenarios and models.




# Key findings

**Regional, sub-regional and national similarities and differences currently exist in the capacity for scenario development and modelling for biodiversity and ecosystem services.** Human resources and the technical skills required for biodiversity and ecosystem services scenario development and modelling are not evenly spread across regions. Differences in capacities for biodiversity and ecosystem services modelling and scenario analyses are most apparent in human resources, infrastructure and technical skills for biodiversity and ecosystem services modelling. External organisations may serve to fill gaps in capacity in nations with smaller economies through the provision of technical and/or financial resources (7.1).

**The ability to develop modelling and scenario analysis for biodiversity and ecosystem services is challenged by a lack of training and human capacity to utilise biodiversity and ecosystem services software and modelling tools.** While many accessible and appropriate software programmes and modelling tools exist, communication of their availability and training in their use is required (7.2).

**Issues regarding the accessibility and compatibility of datasets required for biodiversity and ecosystem services modelling and scenario analysis challenge the ability to develop models and scenarios and to utilise data and model results in assessments.** While many platforms have been developed to serve as repositories of biodiversity and ecosystem services datasets, duplication of effort is common and inconsistencies between formatting and operating standards and lack of complementarity preclude the optimal use of data platforms and their associated datasets in biodiversity and ecosystem services modelling and scenario analysis (7.3).

**The development of biodiversity and ecosystem services modelling and scenario analysis is improving, but tools to incorporate biodiversity and ecosystem services concepts into national and global policy and decision making are underdeveloped and not commonly utilised.** The training and development of human capacity to integrate these tools can enable the incorporation of these tools into policy and decision making. Currently, few scenario tools are available to policymakers that focus on biodiversity and ecosystem services; rather, most scenario analyses are focused on business or economic growth scenarios (7.4).

**A wide range of qualitative and quantitative participatory tools is available to facilitate stakeholder engagement in biodiversity and ecosystem services scenario development.** The involvement of diverse stakeholders and local and traditional knowledge communities in scenario development, including bidirectional communication that recognises and incorporates stakeholder needs into management and policy, is an integral part of successful scenario development (7.5).

## Key recommendations

**The Intergovernmental Platform on Biodiversity and Ecosystem Services (IPBES) Task Force on Capacity Building should consider partnering with existing global programmes, partnerships and initiatives that provide opportunities for networking with respect to human resources and skills development.** For example, the IPBES Task Force on Capacity Building could work with existing Multilateral Environmental Agreements, international organisations and initiatives to provide resources to support joint training initiatives with IPBES to enable participation in the IPBES Work Programme. These partners provide a wide range of training courses, workshops, internships and collaborative projects, including training programmes for trainers. Long-term partnerships could be established with universities in developing and developed countries to train practitioners in tools and software for scenario development and modelling through the development of training courses and mentoring opportunities (7.2.2, 7.2.3, 7.5.3, 7.6.1, 7.6.2).

**IPBES could promote capacity building by providing guidelines and documentation for recommended tools for biodiversity and ecosystem services scenario development and modelling (models, software and databases).** The translation of key documentation into each of the six United Nations (UN) languages and other non-UN languages would contribute greatly to capacity building. These documents should use clear terminology that the users and developers of models and scenarios can understand. IPBES could also develop and support networks and user forums for people to ask questions and interact with other users of models and scenarios, to promote knowledge exchange and the development of capacity within and between regions. Case studies, including access to both model and scenario software and datasets, should be provided to build confidence in using models and scenarios. Intellectual property rights for tools should be determined, and broad access should be taken into account when making recommendations for these models, software programmes and databases (7.2.1, 7.2.2, 7.6.1).

**IPBES should consider identifying standardised global environmental datasets that are required to support IPBES assessments using models and scenarios of biodiversity and ecosystem services.** In cooperation with other partners and donors, IPBES could develop data collection guidelines to build and improve upon environmental datasets that underpin functional relationships between biodiversity and ecosystem services in IPBES models and assessments. Global and regional advisory platforms could be established to develop and adopt global standards and formats for global data and metadata, certify the quality of the datasets, and promote cloud technology with open access to the datasets required for the recommended biodiversity and ecosystem services modelling and scenario tools and software programmes (7.3.1, 7.3.2, 7.6.4).

**The IPBES Catalogue of Policy Support Tools and Methodologies (Deliverable 3d) can build capacity by including guidelines and tools that enable the incorporation of biodiversity and ecosystem services models and scenarios into decision-making processes.** Guidelines and tools are required to identify effective strategies for mainstreaming scenario processes at different geographical scales and to allow their integration into participatory approaches, decision-making processes and public awareness across different policy, planning and management contexts. Identifying and providing capacity for integrating models and scenarios into decision making should take into account the scale — local, regional or global — at which analyses and decision making are made (7.4.1, 7.4.2, 7.5.3, 7.6.1).

**In their efforts to engage and incorporate local and traditional knowledge communities in IPBES assessments, the IPBES Task Force on Indigenous and Local Knowledge Systems should consider the important role that scenarios and models can play in mobilising local and indigenous knowledge.** In particular, it is important to identify and mobilise universities, research institutions and other stakeholders with experience or relationships in the formulation and use of scenarios or models that incorporate indigenous and local knowledge (ILK), as well as to develop networks to share new methods that integrate diverse and multiple forms of knowledge. Scenarios and models can make important contributions to efforts by IPBES to enhance communication between indigenous and local communities, stakeholder groups and local governments, as well as efforts to build the capacity of ILK networks through leadership and educational opportunities (7.4.3, 7.4.4, 7.5.1, 7.5.2, 7.5.4, 7.6.3, 7.6.5).

## 7.1 Introduction

Previous chapters introduced the methodologies for scenario analysis and the modelling of biodiversity and ecosystem services, discussing a wide range of tools that can be used to support IPBES assessment and decision making, as well as other user communities that could benefit from biodiversity and ecosystem services scenarios and models. This chapter reviews the underlying capacity required to support scenario analysis and modelling across a broad range of spatial scales (global, regional and sub-regional) and decision-making contexts.

Key capacity-building objectives regarding scenario analysis and modelling include: to enhance the capacity to develop and use scenarios in assessments, including strengthening human resources and infrastructure; to improve access to and guidelines for user-friendly software tools for scenario analysis, modelling and decision-support systems; to improve regional and national access to and the interoperability of quality standardised datasets; to develop methods for the better incorporation of local data and knowledge; and to develop synergies with existing assessments for data and scenario sharing.

Another key objective is to develop effective strategies for mainstreaming scenario processes at different geographical scales to allow their integration into participatory approaches, decision-making processes and public awareness across different policy, planning and management contexts (Brooks et al., 2014). This chapter discusses the human resources, infrastructure and data accessibility required to enable biodiversity and ecosystem services scenario analysis and modelling at the regional, sub-regional and national scales.

### 7.1.1 Capacity building for biodiversity and ecosystem services scenario development and modelling

The UN Development Programme (UNDP) defines capacity development for environmental sustainability as 'the process through which individuals, organisations and societies obtain, strengthen and maintain their capabilities to set and achieve their own development objectives over time'.

Components of capacity include the skills, systems, structures, processes, values, resources and powers that together confer a range of political, managerial and technical capabilities (UNDP, 2011). Within IPBES, the Task Force on Capacity Building has identified five key capacity-building categories: 1) capacity to participate effectively in implementing the IPBES Work Programme; 2) capacity to carry out and use national and regional assessments; 3) capacity to locate and mobilise financial and technical resources; 4) capacity to access data, information and knowledge; and 5) capacity for enhanced and meaningful multi-stakeholder engagement (IPBES/3/18, Decision IPBES-3/1 Annex I, http://ipbes.net/).

Within the context of biodiversity and ecosystem services scenario analysis and modelling, capacity development includes the human resources and technical capacity required to support scenario analysis and modelling across a broad range of spatial scales (global, regional, sub-regional, national and local) and decision-making contexts (Table 7.1). Data collection skills, such as those of ecologists and taxonomists who collect data related to flora and fauna, as well as of soil scientists and other experts, underpin the databases required to develop scenarios and models.

Capacity building for scenario analysis and modelling also includes the capacity to support the development of effective strategies for mainstreaming scenario processes at different geographical scales. There are many entry points and strategies for developing scenarios and models across scales (Table 7.2), and many entry points for integrating these into participatory approaches, decision-making processes and public awareness across different policy, planning and management contexts (Table 7.3).

**Table 7.1:** Capacity-building requirements for biodiversity and ecosystem services scenario analysis and modelling.

| Activity | Capacity-building requirements |
|---|---|
| Stakeholder engagement | • Processes and human capacity to facilitate engagement with multiple stakeholders, including holders of traditional and local knowledge |
| Problem definition | • Capacity to translate policy or management needs into appropriate scenarios and models |
| Scenario analysis | • Capacity to participate in development and use of scenarios to explore possible futures, and policy or management interventions |
| Modelling | • Capacity to participate in development and use of models to translate scenarios into expected consequences for biodiversity and ecosystem services |
| Decision making for policy and management | • Capacity to integrate outputs from scenario analysis and modelling into decision making |
| Accessing data, information and knowledge | • Data accessibility<br>• Infrastructure and database management<br>• Tools for data synthesis and extrapolation<br>• Standardisation of formats and software compatibility<br>• Human resources and skill base to contribute to, access, manage and update databases<br>• Tools and processes to incorporate local data and knowledge |

**Table 7.2:** Capacity-building objectives, strategies, actions and entry points for developing biodiversity and ecosystem services models and scenarios.

| Capacity-building goal | Strategies | Actions and entry points |
|---|---|---|
| 1. Enhance national and regional networks, individuals and team capacities to carry out scenario exercises within IPBES assessments | • Establish or strengthen regional networks of experts<br>• Update and complement knowledge and skills in scenarios<br>• Improve research capacities of universities and other research and training institutions<br>• Implement biodiversity and ecosystem services scenario and model training | • Map current expertise/capacities (local and regional)<br>• Identify needs<br>• Host regular national and regional training workshops to share methodologies<br>• Host workshops for specific technical aspects<br>• Assist in conducting scenarios within assessments in real settings<br>• Train new and emerging actors in applied settings<br>• Develop curricula relating to ecosystem services and development of scenarios<br>• Involve students and young researchers (e.g., through fellowships) |
| 2. Enhance institutional expertise, particularly on the science–policy interface, for effective adoption of the scenario findings | • Engage stakeholders<br>• Enhance the science-policy interface in support of implementing scenarios<br>• Improve the shared knowledge base<br>• Improve understanding of the decision-making process on the part of the scientific community<br>• Improve capacity for transdisciplinary and trans-sectorial communication | • Establish inclusive assessment governance structure (stakeholders, scientists, policy makers, local organisations or individuals)<br>• Host networking and face to face meetings with multiple stakeholder groups<br>• Promote dialogue and development of visioning exercises with multiple actors (scientists, government officials, policymakers and other stakeholders)<br>• Promote dialogues on scenario approaches to improve the shared knowledge base (including qualitative and participatory approaches)<br>• Train in communication skills |
| 3. Strengthen institutional and organisational structures at all levels | • Assess, revise and develop scenarios and modelling capacities<br>• Enhance the capacity to participate effectively in IPBES assessments<br>• Develop capacity to locate and mobilise financial and technical resources<br>• Establish exchange programme and technical assistance | • Develop plans of actions<br>• Establish institutional partnerships at all scales<br>• Promote IPBES matchmaking to bring together experts, practitioners, mentors, and local knowledge holders with financial resources<br>• Increase cooperation between centres of excellence/institutions<br>• Create common platforms, working groups of diverse knowledge holders including ILK holders |

**Table 7.3:** Capacity-building objectives, strategies, actions and entry points for enabling target groups to use biodiversity and ecosystem services models and scenarios.

| Capacity-building goal | Strategies | Actions and entry points |
|---|---|---|
| Enhance decision making processes at national, regional and global levels by enabling policy makers to better use scenario outcomes | • Map priority areas for capacity needs for users and develop tailor made courses on prioritised thematic subjects | • Identify the expectations and priority areas on capacity needs for users<br>• Develop user friendly and open access prototypes of models and scenarios<br>• Develop simplified modules and courses on focussed thematic subjects relevant for effective decision making<br>• Organise short and tailor made courses on prioritised thematic subjects<br>• Evaluate relevance and effectiveness of the courses and modules and adjust periodically |
| Institutionalise science–policy interface in governance systems for effective adoption and use of the results from models and scenarios | • Encourage institutions responsible for decision making to provide opportunity to develop skill and include such expertise in the team | • Include skilled human resources having understanding on scenarios and models in science – policy interfaces<br>• Enhance trust in models<br>• Develop skills to interpret model robustness, how well models represent biodiversity and ecosystem services processes and reproduce observed behaviours and how sensitive they are to uncertainty<br>• Promote multidisciplinary teams in decision making process<br>• Provide incentives for using science-policy interface in decision making process<br>• Promote the inclusion of use of models and scenarios throughout the policy cycle (see Chapter 2) |
| Strengthen science–policy interface using models and scenarios through human resources development | • Establish curriculum on science–policy interface using models and scenarios at different levels | • Organise tailor made diploma courses for professionals engaged in decision making positions<br>• Initiate the concept of decision-making process using models and scenarios at school curriculum |

### 7.1.2 Current capacity for effectively participating in the development and use of scenarios and models in IPBES assessments

Regional, sub-regional and national similarities and differences exist in capacities to participate in biodiversity and ecosystem services scenario analysis and modelling.

These differences are a reflection of political history, environmental variability, information and communications technology, economic capacity, population size and education, among many other factors (Rodrigues et al., 2010). Differences in capacity are most noticeable when comparing the support infrastructure for scenario analysis and modelling across nations and regions.

Significant differences are apparent when comparing the economic investment priorities of different governments, including the prioritisation of biodiversity and ecosystem services research (Figure 7.1A). Disparities in the authorship of scientific papers on biodiversity and ecosystem services models highlight the cross-regional and national differences, reflecting differences in both human and technological capacities in biodiversity and ecosystem services modelling (Figure 7.1B). Unfortunately, biodiversity-rich countries and regions are not the main contributors to biodiversity and ecosystem services modelling and scenario analysis. Additionally, there are geographical inequalities in access to information on biodiversity and ecosystem services scenarios and models, and to the datasets and software tools used to develop them, as approximated by relative internet usage (Figure 7.1C).

Innovations in biodiversity and ecosystem services models are often supported by government funding through academic and research institutions or through direct funding by government ministries to develop and implement management solutions. However, there is a dependency on external organisations (e.g. environmental non-governmental organisations (NGOs)) to provide technical and financial resources in many nations with smaller economies, with resulting challenges relating to long-term viability and uptake by local stakeholders (Morrison et al., 2010; Horigue et al., 2012; Mills et al., 2015).

There are also cultural differences at local, regional and national scales that need to be recognised in biodiversity and ecosystem services scenario planning processes. These include bias due to a lack of cross-cultural engagement and understanding, and also bias where local or traditional management practices, customary and participatory decision making, and oral knowledge and data gathering are not integrated into policy and decision making. Cultural frameworks also guide taboos about types of management and decision-making frameworks that are acceptable, and acceptable methods of collecting and sharing data. The separation of people and nature can result in discontinuities between local community priorities for biodiversity management, and those of government institutions.

Thematic bias is seen at the ecosystem scale, with biodiversity and ecosystem services models and scenarios more commonly used to support decision making in terrestrial ecosystems than in marine and freshwater ecosystems (FRB, 2013). Socio-economic drivers also result in differing capacity across topical issues, with model capacity biased toward resource-based modelling (e.g. fisheries, forestry and agriculture) and fewer resources allocated to models that have little underlying economic gain. The increased understanding and integration of ecosystem service concepts into environmental policy, and the recognition of ecosystem services concepts in international commitments on platforms such as the Intergovernmental Panel on Climate Change (IPCC) and IPBES are resulting in models that are more integrated and include environmental (e.g. water quality), climate (e.g. coastal inundation, sea level rise, ocean acidification) and biodiversity objectives alongside socio-economic, cultural and community objectives.

Finally, external drivers can influence the use of biodiversity and ecosystem services scenarios and modelling. Political agendas, which vary on temporal scales of political tenures, can provide impetus or hindrance for innovations and decision making, and can also bring instability by causing reversals of existing decisions and environmental commitments (e.g. Australia's 2014 decision to repeal its carbon tax, and the resulting changes in institutional support for climate-related research). National and regional environmental policies often have a topical bias (e.g. toward terrestrial over marine and aquatic policies) that drive funding, data collection and decision making. Similarly, NGOs have research priorities that may result in bias in research agendas, such as a focus on protected area implementation rather than sustainable agriculture or water quality.

With an understanding of historical differences and similarities in capacity for biodiversity and ecosystem services modelling and scenario analysis, future strategies for capacity building can expand on these existing capacities and fill national and regional gaps. In the remainder of this chapter, we present strategies to develop capacity for effective participation in the development and use of scenarios and models in IPBES assessments, to access data, information and knowledge, to integrate biodiversity and ecosystem services models and scenarios into policy and decision-making frameworks, and to ensure meaningful multi-stakeholder engagement.

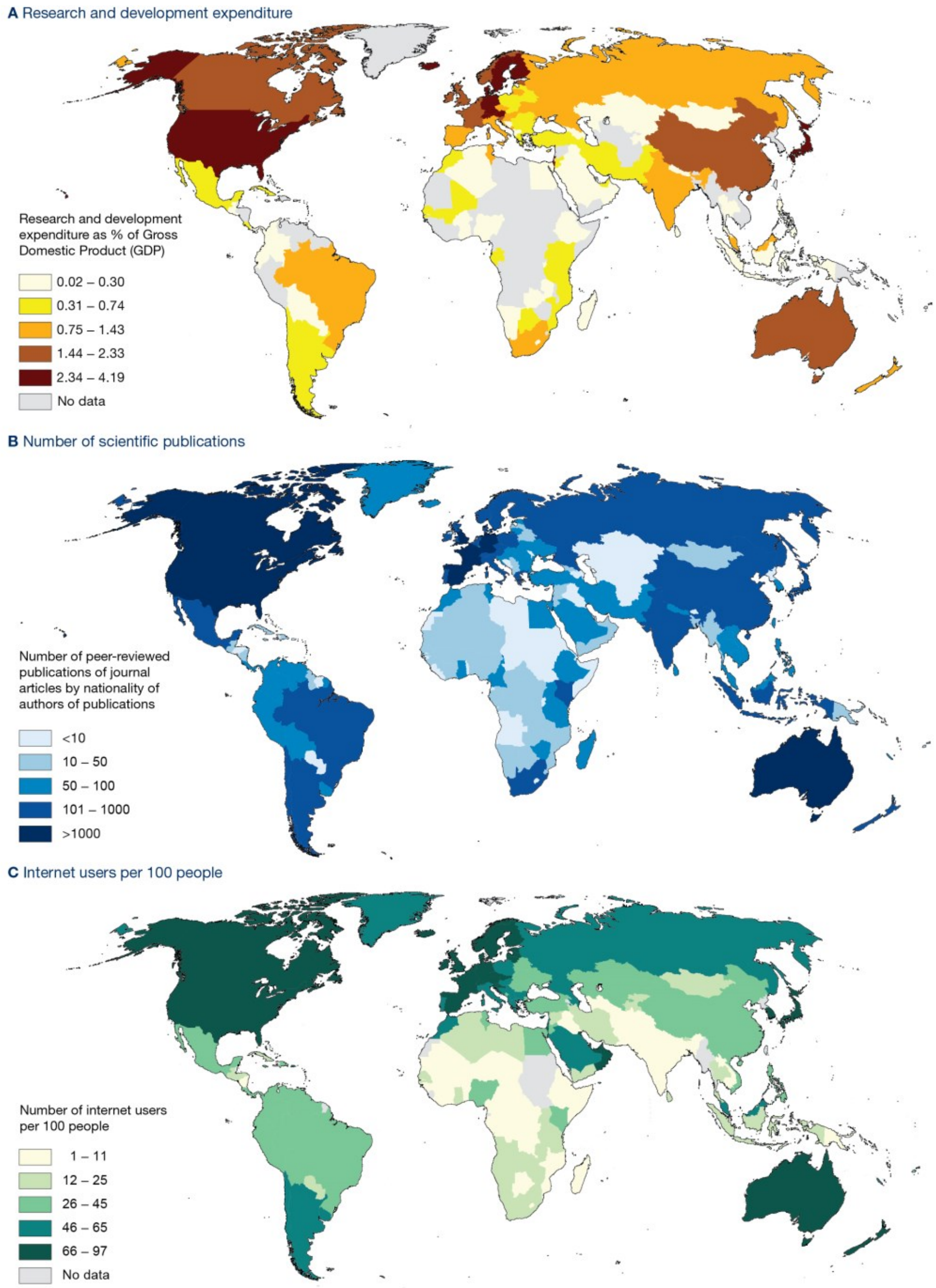



**Figure 7.1:** Regional differences in capacity to support biodiversity and ecosystem services modelling and scenario analysis. A. **Research and development expenditure (RDP as a % of Gross Domestic Product (GDP))**. Current and capital expenditures (both public and private) on creative work undertaken systematically to increase knowledge, including knowledge of humanity, culture and society, and the use of knowledge for new applications. (Data source: UN Educational, Scientific and Cultural Organization (UNESCO) Institute for Statistics, http://databank.worldbank.org/); B. **Peer-reviewed publications of scientific and technical journal articles** based on a search of the ISI Web of Science citation database for all years (1900–current) for the nationality of authors of publications with TOPIC: (ecosystem service*) OR TOPIC: (biodiversity*) AND TOPIC: (model* OR scenario*); C. **internet users** per 100 people. (Data source: World Bank/World Development Indicators, http://databank.worldbank.org/).

## 7.2 Enhancing capacity to effectively participate in the development and use of biodiversity and ecosystem services scenarios and models

It is important, also for the IPBES Work Programme, to enhance people's capacity to effectively participate in the development and use of biodiversity and ecosystem services scenarios and models (Annex 4, IPBES Task Force on Capacity Building). Developing and using biodiversity and ecosystem services scenarios and models requires expertise in various fields, such as ecological processes, modelling, economics, geographic information systems and the social sciences, to contribute to regional, global and thematic assessments. The development of policy-support tools and methodologies to integrate models and scenarios into national and regional decision making requires the expertise of ecologists, social scientists, economists, lawyers and policy analysts. In addition, facilitators with experience in participatory approaches are needed to enable the incorporation of local and traditional knowledge and stakeholder input into scenarios, models and decision-making processes.

### 7.2.1 Technical capacity for effective participation in models and scenarios

Key aspects of the technical capacity required for scenario analysis and modelling include improving access to and guidelines for user-friendly software tools for scenario analysis, modelling and decision-support systems.

There is a clear need for guidelines and documentation on recommended scenario development and modelling tools (models, software and databases) in the six UN languages and other languages where appropriate, using clear terminology that users and developers of models and scenarios can understand. The development and support of networks, workshops and user groups for people to ask questions and interact with other users of models and scenarios could promote knowledge exchange and the development of capacity within and between regions.

Case study examples can help build confidence in the use of scenarios and models for biodiversity and ecosystem services analysis, by providing models, software and actual datasets to allow the development of skills in their use. Visualisation tools included with open access software, such as the CommunityViz geovisualisation tools (http://placeways.com/communityviz/), can assist in exploring modelling software and the implications of different management scenarios. These will enable improvements in the exploration and communication of alternative scenarios and promote more effective planning and management.

The most important aspects for the successful use of biodiversity and ecosystem services models and software tools are accessibility, user-friendliness and the robustness of these tools.

Models can be used individually or combined within scenario analyses to describe relationships between indirect drivers, direct drivers, and biodiversity and ecosystem services, resulting in predictions that relate to nature's benefits to people. The software used in biodiversity and ecosystem services models ranges from commercial applications such ArcGIS and other geospatial software, to specialist tools developed specifically to model ecosystem services (e.g. Integrated Valuation of Ecosystem Services and Trade-offs (InVEST)), to applications for mobile phones such as those created to support the taxonomic identification and geospatial recording of biodiversity records (Table 7.4; reviewed in Bagstad (2013)). There are also models specifically developed to suit local or regional situations.

Intellectual property rights can influence access to both software and datasets used in biodiversity and ecosystem services models and scenarios. While many tools are open source and freely accessible, access to proprietary software can be attained through financial support from funding sources such as the UN, the World Bank and the Convention on International Trade in Endangered Species (CITES). Examples of open source biodiversity software tools include Waterworld and Co$ting Nature (Table 7.4). Other tools, such as Vensim, offer versions that are free for academic use or free for a period of time or with limited functionality to allow people to begin to use the tools. Co$ting Nature provides free web training for their user base and includes links to most global datasets in their TerraSim server; this software also provides the option to upload other databases if better data are available. If computing resources are limited, cloud technologies can be harnessed to allow for adequate processing power to perform models and scenarios using large datasets.

**Table 7.4**: Comparison of accessibility and usability of widely used software for biodiversity and ecosystem service models and scenario analysis (see also Chapter 4, Table 4.3 and Chapter 5, Table 5.4 for detail related to the use of these and similar modelling tools in biodiversity and ecosystem services models and scenario analysis).

| Name | Tool categories | Open source | Documentation and training | Additional software requirements | Data requirements |
|---|---|---|---|---|---|
| **ARIES** http://www.ariesonline.org/ | Stakeholder engagement and outreach; Modelling and analysis; Visualisation; Decision support | Yes | Online documentation; online training via webinars; technical support as needed | No resources required to use basic functionalities (data are included, tool is web-based). Detailed analysis may require data input for region of interest if not already available | Probabilistic and capable of operating in conditions of data scarcity. No data is required for basic analysis, but user data can be input to improve predictions |
| **ECOSERV** http://www.durhamwt.com/wp-content/uploads/2012/06/EcoServ-GIS-Executive-Summary-Only-WildNET-Jan-2013-9-pages.pdf | Modelling and analysis; Decision support | Free for Wildlife Trusts | Online documentation | ArcGIS 10 or higher software | Geospatial data |
| **GLOBIO3** http://www.globio.info/ | Modelling and analysis; Decision support | No | Online documentation, in person training workshops | Specialist training and software required | Unspecified |
| **INVEST** http://www.naturalcapitalproject.org/ | Modelling and analysis; Visualisation; Decision support | Yes | Online user guide, in person and online training and webinars, online forum for troubleshooting | ArcGIS (ArcInfo 9.3 or higher) | Biophysical data (e.g., land cover, topography, precipitation); socio-economic data (e.g., population density, property values, operating costs, market prices of natural resources) |
| **LUCI** http://www.victoria.ac.nz/sgees/research/research-groups/enviro-modelling/ecosystem-service-modelling#lucideveloping | Stakeholder engagement and outreach; Modelling and analysis; Visualisation; Decision support | No | Not currently available | ArcGIS | Geospatial data |
| **SOLVES** http://solves.cr.usgs.gov/ | Stakeholder engagement and outreach; Modelling and analysis; Visualisation; Decision support | Yes | Online user guide and tutorials | ArcGIS 10, 10.1 or 10.2 software | ArcGIS supported formats for geospatial and tabular data |
| **WATERWORLD/ COSTING NATURE** http://www.policysupport.org/waterworld, http://www.policysupport.org/costingnature | Data processing and management; Stakeholder engagement and outreach; Modelling and analysis; Decision support | Free for non-commercial uses | Online documentation, in person and online training | GIS skills useful but not necessary | None - all data supplied |

### 7.2.2 Developing capacity to participate in assessments and the development of policy-support tools and methodologies

Training programmes are an important part of building human capacity to support biodiversity and ecosystem services models and scenarios analysis.

Training programmes should be provided in the most widely used language in a region to support the development of biodiversity and ecosystem services skills (Paulsch et al., 2015). A selection of training programmes relevant to IPBES include those training programmes associated with the UN Environment Programme-World Conservation Monitoring Centre (UNEP-WCMC) (www.unep-wcmc.org/expertise), Massive Open Online Courses (MOOC) (https://www.mooc-list.com/), the Sub-Global Assessments Network (SGA) (http://www.ecosystemassessments.net/network/mentoring-scheme.html), and the International Union for Conservation of Nature (IUCN) Red List training course (http://www.iucnredlist.org/technical-documents/red-list-training). These training programmes perform a wide range of activities, from coursework, student supervision and mentoring of early career scientists, to project placement and capacity building to promote skills in the field of ecosystem assessment. The recently established IPBES Mentoring programme will also mentor early career scientists in developing skills to participate in assessments within the IPBES Work Programme.

Training is also an important component of software applications. Regular courses are run at global, regional and national scales – including through online training courses and webinars – and provide guidance on the use and application of different models and software tools (Table 7.4). Short-term training courses are also often held in association with meetings of scientific societies or through various regional and national projects. For example, projects such as the Climate Change Impacts on Ecosystem Services and Food Security in Eastern Africa (CHIESA) under the International Centre for Insect Physiology and Ecology (ICIPE) sponsored courses to train practitioners in some of the tools (such as InVEST) in biodiversity and ecosystem services scenario analysis and modelling. Short courses and workshops can also be used to provide training in a selection of key biodiversity and ecosystem services scenario and model tools. Regular courses to support the development of biodiversity and ecosystem services skills will enhance the capacity of practitioners and early career researchers, especially those from developing countries, in addition to sharing knowledge and skills and establishing networks across geographical boundaries.

The development and interpretation of scenarios requires the explicit acknowledgement of the interdependencies between system components and the uncertainties associated with ecosystem driver trajectories. To be the most effective for decision makers, an understanding of the different parameters that can produce a range of possible futures is also needed. This 'what if' analysis (Costanza, 2000; Watson and Freeman, 2012) can be considered an extension of a sensitivity analysis, where all inputs are consistently modified against an overarching theme or narrative (Francis et al., 2011). Training in scenario analysis ideally includes detailed documentations of parameters and model inputs (if these are inbuilt in scenarios). In addition, information and training for scenario analysis are optimised when linked to the development of models and software tools.

### 7.2.3 Developing and utilising networks to enhance capacity to implement the IPBES work programme

International environmental governance literature generally conceives of ‘networks’ as the links created by and through social relations in economic, cultural and political domains, with an emphasis on the materiality of the operation and practice of these networks (Bulkeley, 2005). Using this definition to guide the development and utilisation of networks to enhance the capacity for implementing scenarios and models in the IPBES Work Programme can focus attention on the support of various educational and development pathways at a range of interconnected scales.

Many global programmes, partnerships and initiatives provide opportunities for human resource and skills development associated with biodiversity and ecosystem services, through a wide range of training courses, workshops, internships and collaborative projects. Long-term partnerships with universities in developing and developed countries can provide practitioners with training in tools and software for scenario analysis and modelling, through the development of short courses and the establishment of MPhil/research fellowships. For example, the Oppla network, currently being developed with European Union funding, will provide facilities to support communities of science, policy and practice for ecosystem services and natural capital, including training courses, guidance documents and networking opportunities.

Similarly, another way of enhancing people’s capacities to use tools is the reinforcement and support of the existing regional infrastructure for modelling biodiversity and ecosystem services. Such infrastructure is already present in many places, but often lacks funding for training or is not well known. By developing a relationship with the agencies and institutions that already have some ecosystem services modelling capacity, it may be possible to implement a ‘train the trainer’ programme that could exponentially enable capacities.

The creation of networks and user forums that include scientific communities, stakeholders, decision makers and policymakers can enable feedback at every stage of model development, including the evaluation of scenario and model outputs using empirical observations.

Such networks and forums are useful for people to ask questions, interact with other users, and exchange knowledge. Communities of practice around specific modelling and scenario tools, such as Marxan and Ecopath with EcoSim (EwE), can build capacity in software use, serve international networks of users, and answer queries ranging from software applications to dataset requirements related to the software.

There is also a need to build communities of practice around broader aspects of modelling and scenarios. International programmes such as the Natural Capital Project (http://www.naturalcapitalproject.org/) and The Economics of Ecosystems and Biodiversity (TEEB) (http://www.teebweb.org/) provide such networks.

## 7.3 Improving capacity to access data, information and knowledge

Datasets are an essential contribution to our understanding of biodiversity and ecosystem services.

Biodiversity datasets were used to establish the fact that governments missed their targets for reducing the rate of biodiversity loss by 2010 (sCBD, 2010). Although the rate of loss was significantly reduced relative to potential biodiversity losses in the absence of existing conservation efforts (Hoffmann et al., 2010), progress toward the 2020 Aichi biodiversity targets has been limited (sCBD, 2014). These analyses were interpreted through existing datasets by utilising biodiversity and ecosystem service modelling and scenario development processes (e.g. Sala et al., 2000; Leadley et al., 2010; Pereira et al., 2010; Pereira et al., 2013). One of the many reasons for this global failure was the shortage of comprehensive indicators and associated accessible data (Butchart et al., 2010; sCBD, 2010). To create appropriate policies to protect biodiversity, it is essential that we understand how humans benefit from biodiversity, how species interact, and how they might respond to changes in pressures, both natural and man-made (Mace et al., 2010; Brooks et al., 2014).

### 7.3.1 Developing capacity to gain access to data, information and knowledge managed by internationally active organisations and publishers

Realising the importance of data, many global, regional and national initiatives have begun to archive different forms of data for use in various decision-making processes (Table 7.5; MA, 2005a; Chettri et al., 2008; Yahara et al., 2014; Viciani et al., 2014). This is true even at the global level, where multilateral environmental agreements such as the UN Framework Convention on Climate Change (UNFCCC), the Convention on Biological Diversity (CBD), the Convention on Wetlands of International Importance (RAMSAR), the Convention on International Trade in Endangered Species (CITES), the UN Convention to Combat Desertification (UNCCD) and the Millennium Development Goals are supported by a range of primary and secondary data both at national and global levels to reach common conservation and development goals. The extensive use of global and regional datasets is also evident in the progressive and refined reporting in the IPCC Fourth Assessment Report (IPCC, 2007) and the IPCC Fifth Assessment Report (IPCC, 2014).

**Table 7.5:** Types of platforms that support model and scenario datasets.

| Type of platform | Scale | Examples |
|---|---|---|
| Multilateral environmental agreements and Biodiversity-related Conventions | Global, regional | Convention on Biological Diversity (CBD), Convention on International Trade in Endangered Species (CITES), Convention on Migratory Species (CMS), Convention on Wetlands of International Importance especially as Waterfowl Habitat (Ramsar Convention), United Nations Convention to Combat Desertification (UNCCD) |
| International Government Organisations (IGOs) | Global, regional | Food and Agriculture Organisation (FAO), Global Environment Facility (GEF), Global Biodiversity Information Facility (GBIF), International Union for Conservation of Nature (IUCN), United Nations Environment Programme (UNEP) |
| Regionally hosted datasets | Regional, sub-regional | Pan-European Species-directories Infrastructure (PESI), Arctic Biodiversity Data Service, ASEAN Biodiversity Information Sharing Service (BISS), European Landscape Convention, The Alpine Convention |
| Global thematic datasets | Global, 'thematic' | Ocean Biogeographic Information System (OBIS), MarineBio Species Database (MarineBio), the Global Invasive Species Database (GISD), BirdLife International, FishBase, HerpNET, Integrated Botanical Information System (IBIS), Integrated Taxonomic Information System (ITIS) |
| Others | All scales | Global Biodiversity Informatics Outlook (GBIO), World Biodiversity Database (WBD), Natura 2000, NatureServe |

Parties to such conventions are obliged to develop clearing housing mechanisms with established national-level accessible datasets. These practices have significantly contributed to dataset development and accessibility. More extensive and accessible datasets are anticipated to improve the accuracy and relevance of biodiversity and ecosystem services models and scenarios, as well as the uptake of these tools in environmental assessments. Some promising efforts relating to the development of global biodiversity databases include the Encyclopedia of Life (Parr et al., 2014), the IUCN Red List of Threatened Species (http://www.iucnredlist.org/) and Key Biodiversity Areas through the Integrated Biodiversity Assessment Tool (Harris, 2015), and the Global Biodiversity Information Facility (GBIF) (Robertson et al., 2014; Hjarding et al., 2015). The GBIF portal has made significant progress in providing access to over 500 million published digital species records, of which about 80% are global georeferenced data records (Figure 7.2). Efforts have also been made to develop thematic datasets on forests (Gilani et al., 2015; Pfeifer et al., 2014), wetlands (Lehner and Doll, 2004; Chaudhary et al., 2014) and mountain ecosystems (Chettri et al., 2008; Guralnick and Neufeld, 2005; Gurung et al., 2011).

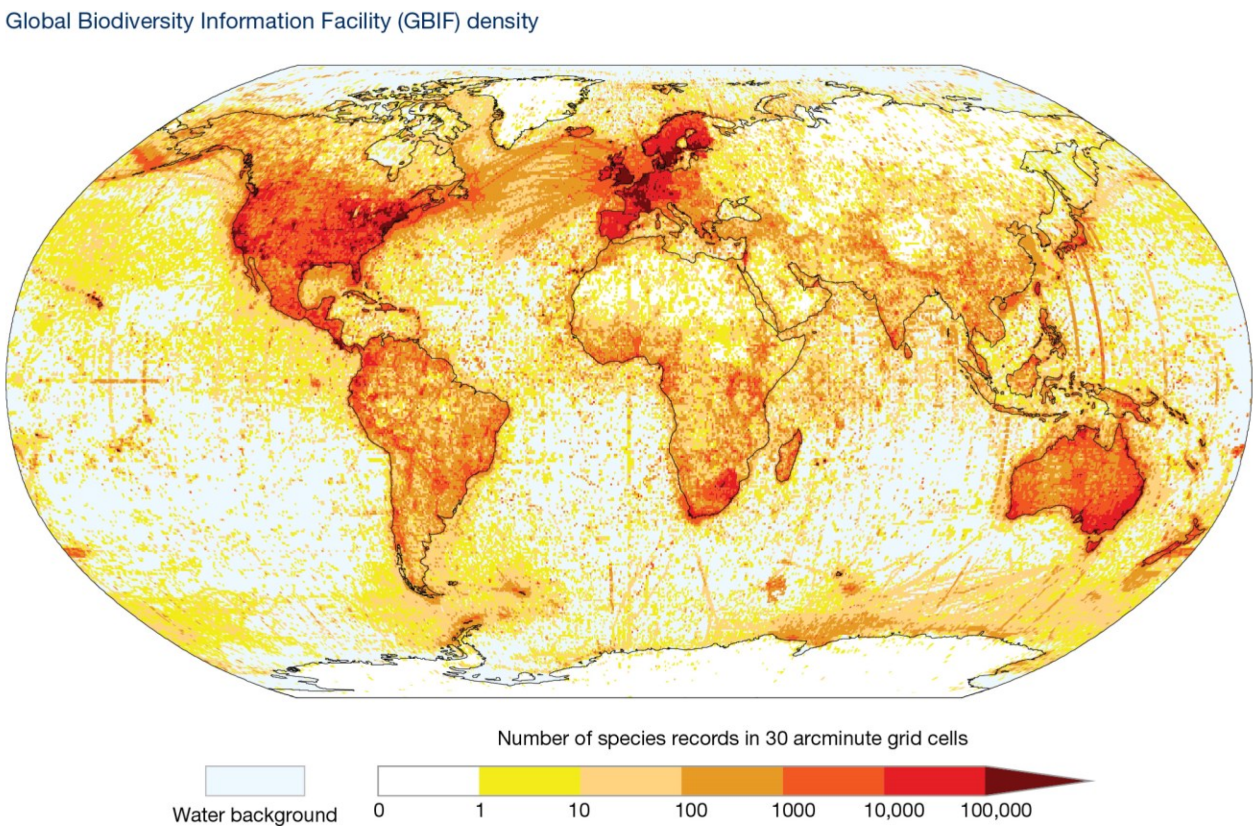



**Figure 7.2:** Density of georeferenced species occurrence records published through GBIF up to December 2015. The top ten contributing countries of georeferenced data include the United States of America, Sweden, the United Kingdom, Australia, the Netherlands, Germany, France, Finland, Norway and Spain. (Modified from http://www.gbif.org/occurrence).

As new technologies and scientific approaches evolve, the modelling of both new and historical datasets can provide an enhanced understanding of the role of biodiversity and ecosystem services in human health and well-being (Pimm et al., 2014). However, this can only happen if we are able to enrich, maintain and use high quality data effectively (GBIF, 2013), for example by carrying out data quality checks to resolve issues of georeferencing and taxonomy in many biodiversity databases. These data quality standards support data archiving in a structured and standardised form to enable a diversity of uses, creating new opportunities for research and applications, and supporting biodiversity-related policymaking. The integration of biodiversity and ecosystem services datasets into innovative modelling tools can enable understanding of scenario trends and projections, and serve as a building block for future conservation and development goals.

Five broad groups of issues are relevant to the access to and incorporation of data into biodiversity and ecosystem services models and scenarios, including intellectual property rights (Arzberger et al., 2004). These are:

1. Technological issues: broad access to research data, and their optimal utilisation, requires an

appropriately-designed technological infrastructure, broad international agreement on interoperability, and effective data quality control (Table 7.6);

2. Institutional and managerial issues: while the core open access principle applies to all science communities, the diversity of the scientific enterprise suggests that a variety of institutional models, intellectual property rights and tailored data management approaches are most effective for meeting the needs of researchers;
3. Financial and budgetary issues: scientific data infrastructure requires continued, dedicated budgetary planning and appropriate financial support. The use of research data cannot be maximised if access, management and preservation costs are an add-on or after-thought in research projects;
4. Legal and policy issues: national laws and international agreements directly affect data access and sharing practices, despite the fact that they are often adopted without due consideration of the impact on the sharing of publicly-funded research data or on intellectual property rights;
5. Cultural and behavioural issues: appropriate reward structures are a necessary component for promoting data access and sharing practices. These apply to those who produce and those who manage research data.

**Table 7.6:** Technical requirements to improve data quality and compatibility.

| Issue | Specific technical requirements |
|---|---|
| Data quality | • Documentation of uncertainties surrounding data collection and measurement error<br>• Authenticity and integrity of data source<br>• Security against loss, destruction, modification, and unauthorised access<br>• Taxonomic resolution and revision (e.g., Nativi et al., 2009).<br>• Interoperability across the temporal and geospatial scales (e.g., Edwards et al., 2000) |
| Data format | • Compatibility with multiple analytical, reporting and publishing options<br>• Compatibility for both spatial and temporal analysis as well as with available software<br>• Geospatial grids and projections appropriate to geographic region and scale, including altitude<br>• Use of commonly recognised and widely used data standards such as Darwin core (e.g., Wieczorek et al., 2012) |
| Data interoperability | • Compatibility of technical standards and protocols across software packages and data management organisations to ensure the access and usability of data |
| Accessibility | • Open access to data and software<br>• Ability to combine data from multiple sources<br>• Comprehensive documentation of datasets |
| Flexibility | • Flexibility to incorporate data management innovations<br>• Flexibility to integrate across disciplines, scales and ecosystems |
| Long-term sustainability | • Financial sustainability to maintain infrastructure including publishing platforms and data hubs<br>• Technological backups of both data and platforms |

## 7.3.2 Developing capacity to enhance multidisciplinary and cross-sectoral collaboration at national and regional levels

Existing data collection and management practices could be improved, with an emphasis on data quality, interoperability, and the institutionalisation of data management processes through short-term and long-term strategies.

Data collection and management have a low priority, leading to the limited representation or participation in the global database development discourse. The vast amount of information available amongst traditional and indigenous peoples and their fading knowledge has not been properly documented and archived. Also, many of the existing global datasets, such as that for forests used in the

History Database of the Global Environment (HYDE) (Klein Goldewijk et al., 2011), have a coarse resolution and do not capture the fine-scaled picture of varied landscapes such as that of mountains or small wetlands and fragmented forests (Sharma et al., 2010; Pfeifer et al., 2014; Svob et al., 2014).

The existing datasets maintained by secretariats of multilateral agreements such as UNFCCC, CBD, RAMSAR, the global commons for bioinformatics such as GBIF and the IUCN Red List, and other datasets maintained by developed countries, do not show complementarity to each other and duplication of work is prominent. Geospatial datasets for the same location may use different geospatial projections, making datasets incompatible (e.g. the numerous geospatial projections available for the Antarctic region and lack of consistency in usage for Antarctic datasets). In addition, taxonomic inconsistencies, the provision for interoperability among the existing datasets, and the duplication of efforts in generating datasets and developing a database infrastructure among biodiversity research communities are introducing greater complexity into the database management domain rather than contributing to its resolution.

The openABM project (openABM.org) provides a useful example of a general model database for biodiversity and ecosystem services models and scenarios. The Centre for International Earth Science Information Network (CIESIN) has assembled multiple datasets to make it easier for modellers to find data. Improving the accessibility, interconnection and metadata of data related to ecosystem service models and scenarios can increase the ease with which models can be created.

A number of capacity-building strategies can result in an increased capacity to use geospatial databases and analytical and visualisation tools for the rapid production of and access to information products (Table 7.7).

**Table 7.7:** Short-term and long-term strategies to address gaps in data collection and management strategies to support biodiversity and ecosystem services modelling.

| Short-term strategies (1-2 years) | Long-term strategies (3-20 years) |
|---|---|
| Incorporate SWOT (strengths, weaknesses, opportunities and threats) analysis | Establish/improve data base infrastructures (portals) and accessibility |
| Prepare database development and management outlook at national and regional levels | Strengthen regional network and cooperation (particularly in nations without culture of data sharing) |
| Develop database catalogue and identification of gaps | Establish global and regional advisory platforms to certify the quality of the datasets in conformity with the adopted standards |
| Provide thematic modules for capacity development (training resources available online) | Link results with policy development process |
| Provide training for database developers and users | Ensure mechanisms exist such that datasets are updated when new information is available |
| Promote data sharing and user policies | Ensure financial sustainability |
| Develop guidelines for habitat assessment to allow approximation of ecosystem services based on land-cover/biotypes, and guide data collection to validate model functional relationships | Develop priorities to enlarge data coverage of global datasets |
| Develop tools for down-scaling of common databases (e.g., GBIF, climate models) | |
| Develop queries to enable dataset transformations of popular global indices to seasonal-monthly-daily scales; spatial queries to enable regional, national or local scale analysis of global datasets | |
| Develop products using new technologies such as Android and iOS applications (e.g., Aichi Indicators Exploration application) and E-books (e.g., E-Handbook of the Convention on Biological Diversity) to communicate information from IPBES | |

## 7.4 Integrating scenarios and models into policy and decision making

### 7.4.1 Capacities required to integrate biodiversity and ecosystem services models and scenarios into policy and decision making

A scenario provides a basis that allows decision units (governments, agencies) to reflect on how changes in developments beyond their immediate spheres of influence, for example in biodiversity and ecosystem services, may affect their decisions. Effective scenario building and model construction require expertise in several fields including management, development, ecology (terrestrial or marine), climate change, culture, agriculture, economics and mapping, depending on the subject at hand (McKenzie et al., 2012).

Chapter 2 identifies the primary impediments to the widespread use of models and scenarios in decision making as a lack of trust in modellers, models and scenarios; a lack of understanding and technical knowledge among decision makers preventing them from understanding outputs and appreciating the positive role that models and scenarios can play; a lack of decision support, modelling and scenario analysis skills relative to the number of policy design and implementation challenges; a lack of willingness on the part of some modellers to engage fully in real-world decision problems and develop and communicate in a non-technical way; a lack of willingness of modellers to engage in participatory processes involving other knowledge traditions; and a lack of transparency in approaches to modelling and scenario development.

Capacity building for decision making based on biodiversity and ecosystem services scenarios and models requires strengthening or developing long-term, relevant, transdisciplinary expertise, institutions, and organisational structures to carry out scenario exercises and develop and use models in IPBES assessments (Ash et al., 2010). This capacity will allow decision makers to act on the findings of biodiversity and ecosystem services models. The purpose of using scenarios and developing storylines is to encourage decision makers to consider certain positive and negative implications of different development trajectories (MA, 2005a). Strategies for mainstreaming scenarios and models into decision-making processes across scales (national, regional and global) and across different policy, planning and management contexts within the framework of IPBES are summarised in Tables 7.1 and 7.2.

The key steps towards mainstreaming scenarios and modelling into the science-policy interface may involve:

(1) Engaging the policymakers and all other stakeholders from the beginning;
(2) Developing relevant biodiversity and ecosystem services scenarios and models that are easily understandable;
(3) Translating results into policymakers' and stakeholders' language;
(4) Using just 'sufficient' data (not too much) to convey a clear message;
(5) Using precise and credible information for biodiversity and ecosystem services scenarios and models.

### 7.4.2 Strategies to mainstream scenarios in the science-policy interface

At the national scale, most governments recognise the social role of ecosystems and their biodiversity due to their influence on human health and quality of life, in addition to their contribution to social and economic development through the supply of essential ecosystem services. This emphasises the socio-cultural and economic value of ecosystem services and the importance of their inclusion in policies. As an example, the failure to meet the 2010 biodiversity targets (sCBD, 2010) stimulated a set of new future targets for 2020 (the Aichi biodiversity targets). As highlighted by Perrings et al. (2011), the first strategic goal to meet the 2020 targets is to 'address underlying causes of biodiversity loss by mainstreaming biodiversity across government and society'. The Millennium Ecosystem Assessment (MA) has shown that there is no clear institutional response to address these underlying causes (indirect drivers of change), and new sets of responses are necessary to meet the 2020 targets. This requires structural changes to recognise biodiversity as a global public service as well as to integrate biodiversity conservation into policies and decision frameworks (Rands et al., 2010) at local, regional and national scales. Biodiversity and ecosystem services scenarios and models can help to fill this gap, but there are currently very few scenarios that focus on biodiversity and ecosystem services and that are suitable for the purposes of policymakers and decision makers. Costanza et al. (2015) reviewed various scenarios at the global and national scales (i.e. Australia), but most of the scenarios were related to business or the economy, not to biodiversity and ecosystem services.

The ongoing accelerated changes in economic, social and environmental aspects require flexible policies. Policy is subject not only to a political process but also to urgent or sudden calls for decisions, sometimes before any scientific result is available (Scheraga et al., 2003). The complexity of ecosystems and their services demands reliable data and analysis for policy decisions (UNEP, 2012; Swanson and Bhadwal, 2009) (Figure 7.3). In addition, there is a growing need for scientific knowledge that is understandable across diverse stakeholder groups.

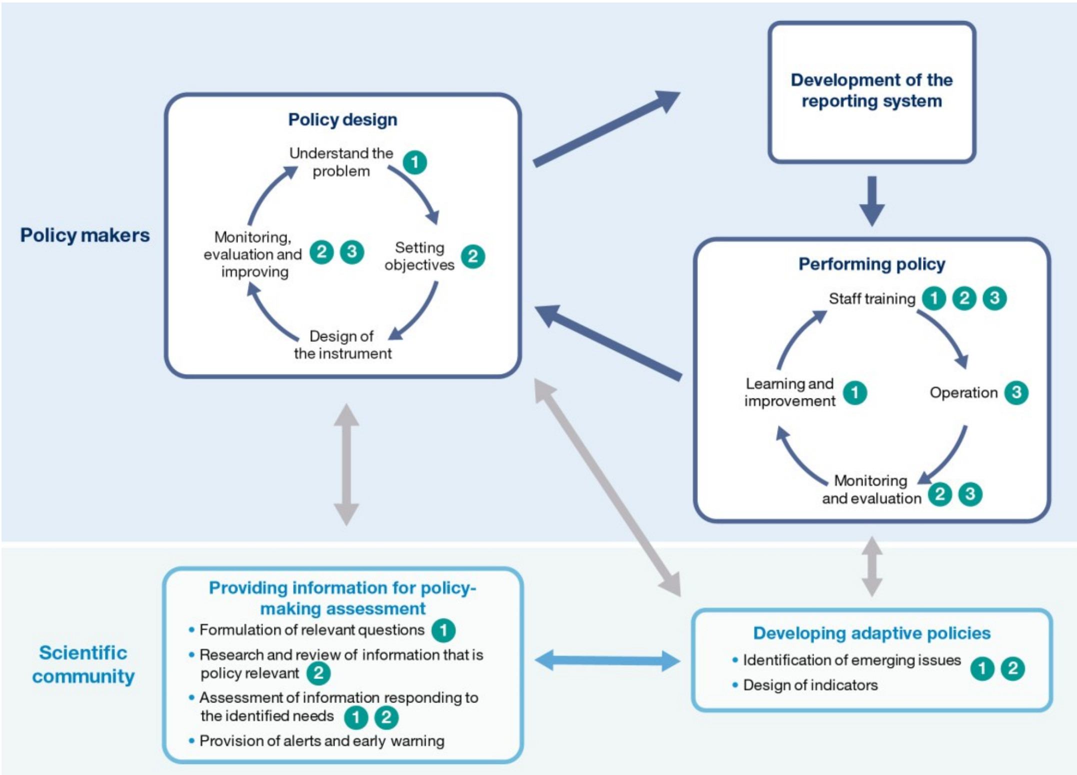



**Figure 7.3:** Linkages between policymakers and the scientific community and the need for scenario analysis capacity (red circles indicate capacity-building objectives as referenced in Table 7.2).

There are at least two different ways in which scenarios and models may be useful for mainstreaming biodiversity and ecosystem services into policy at several scales of decision making:

- 'Scenarios based on models' could be developed to project possible futures where there is a greater degree of certainty in data. For example, population models could be used to develop scenarios on the use of ecosystem services in a particular region.
- 'Models based on scenarios' could be used to project possible future options. A model can use different scenarios to suggest various options that may occur in the future. For example, a model can project variations in values of ecosystem services over time based on the current use of ecosystem services, as in the scenarios-based models used in the United Kingdom (Haines-Young et al., 2014).

Either of the two methods mentioned above can be applied to project long-term impacts for future decision making. However, the second approach could be more appropriate in relation to biodiversity and ecosystem services assessment, given the intangible nature of many ecosystem services and the uncertainty in biodiversity and ecosystem services data. Experts, locals and other stakeholders can then apply their common judgment to predict future alternatives.

To mainstream scenarios and models into policy and decision making, it can be valuable to include people's well-being, the economy and status and trends in biodiversity and ecosystem services as important domains in any biodiversity and ecosystem services scenario, to appropriately dialogue with policymakers. An example of a possible 'biodiversity and ecosystem services approach' is presented below (Figure 7.4):

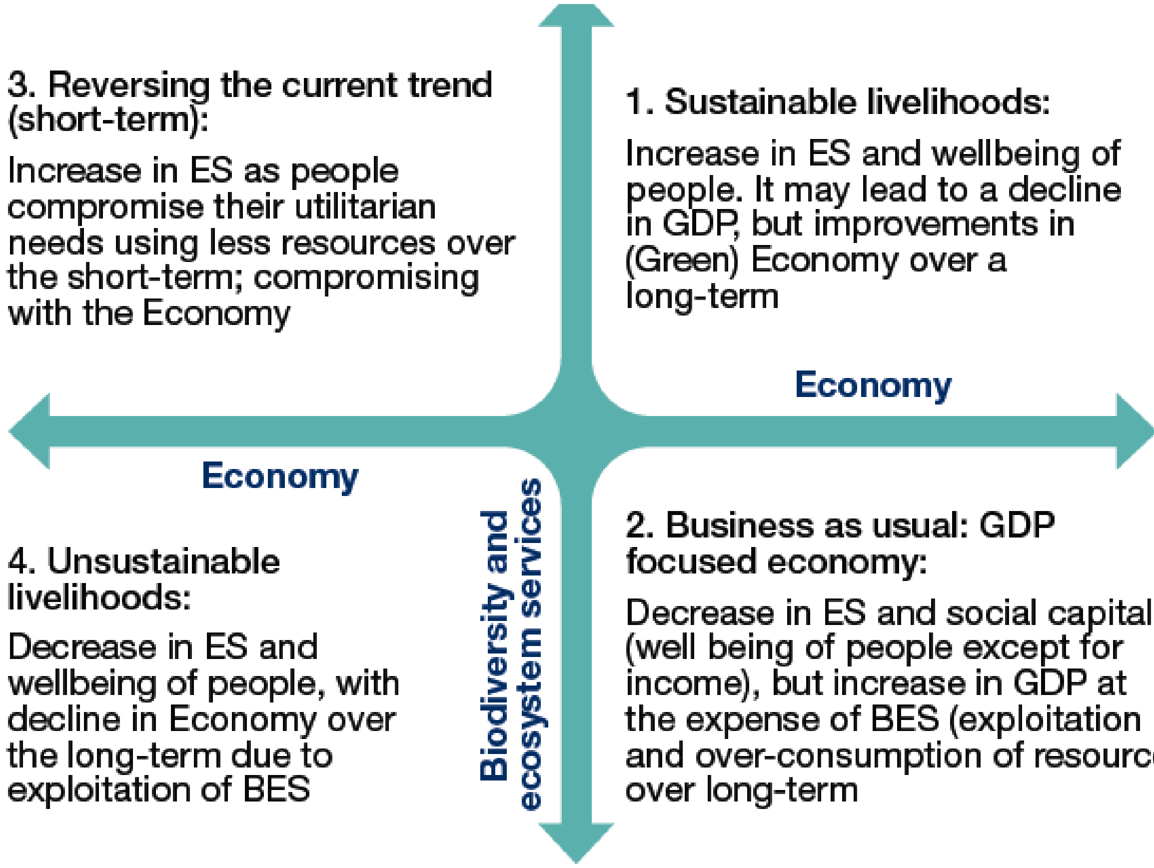



**Figure 7.4:** Example of biodiversity and ecosystem services scenarios linking biodiversity and ecosystem services with economy (focus on GDP and human well-being). Benefits to biodiversity and ecosystem services increase along the vertical axis; benefits to the economy increase along the horizontal axis.

Each type of scenario mentioned in Figure 7.4 can also include a study of the impacts of changes in biodiversity and ecosystem services in the long term on: i) the government (development and policy sector), ii) natural resources (capital), and iii) social values (capital).

A combined scenario planning and modelling approach can be useful for policy decision makers to comprehend various values and changes that may occur in the benefits to humans from biodiversity and ecosystem services in the long term. However, it is important when working with local or indigenous communities to develop scenarios that match people's values. This is one major difference compared with the modelling approach, in which pre-developed models are applied without the inclusion of local values. Scenarios can help explore options from local perspectives, and can accommodate local knowledge on the benefits of biodiversity. This may prove very useful for IPBES assessments, in demonstrating the role of ecosystem services in people's well-being beyond the tangible measures, and in making a significant contribution to bridging the gap between local knowledge and policy decision making.

### 7.4.3 Recognition of the interdependence of knowledge systems, including traditional knowledge, to inform biodiversity and ecosystem services models and scenarios

'Traditional and local knowledge' refers to knowledge and 'know-how' accumulated by regional, indigenous or local communities over generations that guides human societies in their interactions with their environment (IPBES/2/17, http://ipbes.net/). The IPBES Conceptual Framework clearly recognises the importance and interdependence of knowledge across multiple systems (local, scientific, technical, educational and traditional) (IPBES/2/17, http://ipbes.net/), and that an understanding of these complex knowledge systems is necessary to determine system feedbacks within models and scenarios. Folke et al. (2002) and Tengö et al. (2012) highlighted the importance of such knowledge systems for building resilience in a world of uncertainties.

Traditional and local knowledge offers a vision of the world based on a different knowledge system, which provides a new perspective for defining relationships between people and the environment and for constructing 'another possible world' (Leff, 2011). The development of scenarios or models based on cultural understanding (a common vision established in the UN Strategic Plan for Biodiversity 2011–2020) is a critical aspect of scenario analysis and modelling. For indigenous and local communities, environmental management decisions are intrinsically tied to culture and way of life, and their knowledge can enrich and inform scenarios and models (Feinsinger, 2001). However, these systems are often quite complex due to multiple interactions between people and their environment. The main problem with such complex systems is the limited skills available to understand, predict and control socio-ecological systems (Pilkey and Pilkey-Jarvis, 2007; Roe and Baker, 2007; Eddy et al., 2014). There is therefore a need to develop an integrated system of conventional scientific and traditional knowledge, for which decision making must engage with the most relevant users (Cortner, 1999; Bocking, 2004; MA, 2005b, MA, 2005a).

Biodiversity and ecosystem services scenarios and models must integrate key aspects of local knowledge, including feedbacks between different scales and knowledge systems.

The co-design and co-production of necessary knowledge in the process of modelling and building scenarios will strengthen human capacities. To develop effective biodiversity and ecosystem services scenarios and models for decision making, diverse forms of local knowledge must come together by transcending spatial and temporal scales. The dialogue of knowledge can form the platform for scenarios and modelling across the scientific interchange, to strengthen the validation and the co-production of knowledge (Figure 7.5). This dialogue can integrate knowledge and world views from local and indigenous perspectives, including civil society, scientific experts, private and economic sectors, and the government. In this process, knowledge is achieved through a combination of rights, obligations and responsibilities, resulting in the integral, just and sustainable management of resources (Pacheco, 2013).

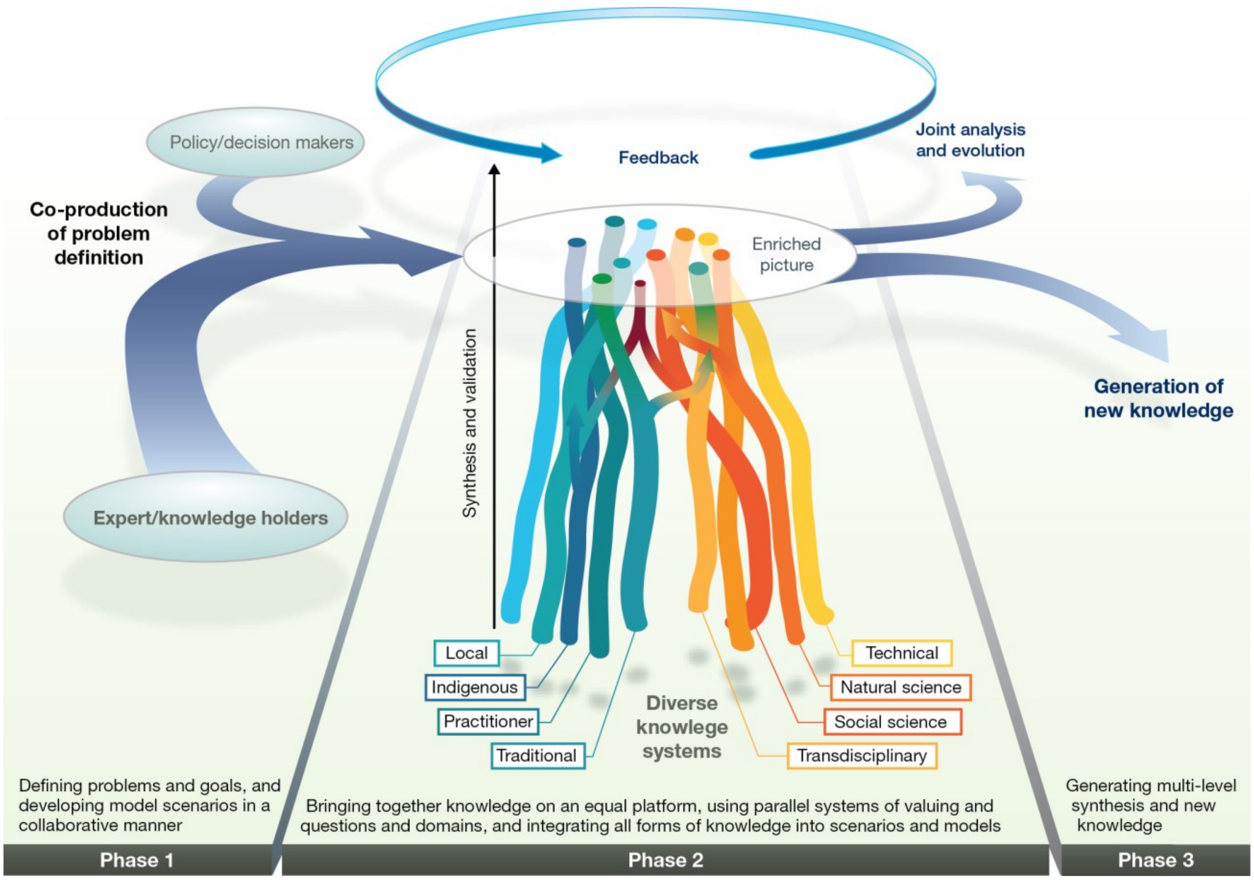



**Figure 7.5:** Conceptual diagram on the integration of local knowledge for developing biodiversity and ecosystem services scenarios and models for decision making (Modified from Tengö et al., 2014, DOI: 10.1007/s13280-014-0501-3, http://creativecommons.org/licenses/by/4.0/). Illustration of how local knowledge of biodiversity and ecosystem services can be integrated throughout all phases of policy and decision making, resulting in the creation of new knowledge that can be used in the development of biodiversity and ecosystem services scenarios and models, and improved policy and decision making.

## 7.4.4 Mechanisms to include indigenous and local knowledge in scenario analysis and modelling

To incorporate traditional knowledge systems into scenario analysis and modelling, the key mechanisms are to integrate knowledge and to enhance participation and dialogue between actors at national and regional scales.

Some key aspects to develop efficient mechanisms for integrating traditional knowledge into biodiversity and ecosystem services scenarios and models are:

1. Develop a good understanding of indigenous knowledge systems and the ability to translate and

integrate this knowledge, where possible, into conventional knowledge systems;

2. Study beyond the set boundaries to embrace the holistic perspectives of living that are embedded in many indigenous knowledge systems; this applies in particular to practitioners in conventional (academic) knowledge systems;
3. Develop a 'common' (integrated) knowledge base through shared traditional and conventional knowledge systems (e.g. a set of indicators);
4. Apply a transdisciplinary approach to the role of biodiversity and ecosystem services in terms of people's livelihoods (well-being), where the MA framework could be useful (but with local modifications);
5. Engage 'effectively' with local and traditional societies from the earliest possible stages of scenario and model development.

Some examples of integration mechanisms include adaptive co-management, participation and ongoing collaboration with traditional and local societies (Folke et al., 2002). Adaptive co-management incorporates traditional and conventional scientific knowledge and encourages participation and collaboration amongst all stakeholders (Paulsch et al., 2015). It is critical to effectively engage with local and indigenous communities and their knowledge from the first stage of planning scenarios in order to allow co-definition of the problem, to increase trust and understanding between participatory stakeholders, and to reduce uncertainty in the scenarios (Peterson et al., 2003). The long-term success of a particular scenario will depend on cooperation among various stakeholders in scenario refinement, testing and iterations, to ensure acceptance for evaluating policies and informing decision making. Effective engagement with traditional and local societies can therefore be key to the development of appropriate biodiversity and ecosystem services scenarios.

The incorporation of traditional knowledge is a process that goes hand-in-hand with the empowerment and strengthening of local communities, and is directly related to Aichi biodiversity target 19. One method for incorporating traditional knowledge is to develop an integrated set of biodiversity and ecosystem services indicators that are based on scientific and traditional knowledge. At a local scale, indicators that include or have links to local and regional traditional knowledge systems will contribute better to collaborative involvement and enhance socio-ecological scenarios and models (IUCN, 2006). Robb et al. (2014) found that the implementation of locally-based indicators in biocultural conservation can be used to integrate local Māori knowledge and conventional academic science. Another example is the Sub-Antarctic Biocultural Conservation Programme, conducted at a local scale, and the National Programme of Conservation and Sustainable Utilization (PNCASL) for the caiman (*Caiman yacare*) in Bolivia, as presented in Box 7.1.

**Box 7.1:** The incorporation of indigenous and local knowledge (ILK) in the management and conservation of *Caiman yacare* (a crocodile species) in Bolivia

Bolivia's National Programme of Conservation and Sustainable Utilization (PNCASL) for the customary harvest and conservation of caiman (*Caiman yacare*) illustrates a case study of the successful integration of ILK into biodiversity models to inform policy options (Llobet et al., 2004; Van Damme et al., 2007; Campos et al., 2010). Previously, harvest quotas were estimated based on broad-scale estimates of abundance from scientific surveys, with substantial variation between regions. The annual assignment of local harvest quotas was estimated across the 'Scientific Authority' based on random counts of relative abundance. Following the increasing engagement of local communities in PNCASL, new biological, socio-economic and cultural indicators of species health and abundance were developed and trialled. These included both biological indicators (based on models of the species) and socio-economic and cultural indicators of species health. One of the first trials took place in the Indigenous Territory and National Park Isiboro Sécure (TIPNIS), where local knowledge was initially the most reliable source on the status of *Caiman yacare*. Here, traditional knowledge on the status of caiman was incorporated into the development of robust indicators to inform resource quotas for customary harvest within this protected area. Traditional resource users participated in workshops where they defined concepts, harmonised criteria and conceptualised traditional knowledge of caiman habitats and territories into spatial maps. Population abundance was measured by scientific researchers, comparing estimates using both scientific techniques and indigenous techniques suggested by the communities (Aguilera et al., 2008). Models for estimating population abundance were adapted to make use of indigenous techniques of estimating caiman abundance and to incorporate qualitative indicators such as individuals' perceptions of changes in caiman abundance, for example accounting for information from statements such as 'there are a lot more caiman than before'. The process was repeated with communities across the TIPNIS territorial region, using this integration of knowledge systems and harvest estimates developed from local knowledge, and fortified with scientific concepts and criteria (e.g. sizes of hunt allowed) that were internalised by the local communities. This integrated process yielded a combined caiman population estimate for the protected area based on local knowledge. This estimate was used to develop a national-scale predictive model of abundance, which then informed national, regional and local policy options for improving the sustainable management of caiman harvest (Aguilera et al., 2008). Resulting management plans for indigenous territories and protected areas have been recognised as contributing to increases in caiman abundance in areas where they had been locally depleted and in reducing illegal hunting. Furthermore, this programme has resulted in benefits to local people, both through the conservation of caiman, and in supporting customary harvest levels that provide economic benefits to local people (Aparicio and Ríos, 2006; UNEP-WCMC, 2013). CITES removed restrictions on the import of wild caiman from Bolivia in 1999 and records a positive caiman status since 1999, which was re-confirmed in 2006 (UNEP-WCMC, 2013). The IUCN/SSC Crocodile Specialist Group has also confirmed a good status of wild populations of *C. yacare* (Larriera et al., 2005; Campos et al., 2010).

Indigenous knowledge can also be integrated in biodiversity and ecosystem services scenarios and models by understanding and evaluating the role of biodiversity and ecosystem services in people's well-being where it can also inform economic theory. This necessitates a need to develop and apply a holistic perspective of well-being for incorporating ecosystem services. Sangha et al. (2011) evaluated the role of ecosystem services from tropical rainforests in indigenous well-being in North Queensland, applying the MA approach (Figure 7.6). Each ecosystem service/well-being link highlighted the importance of an ecosystem service in terms of the well-being of indigenous people that could be used in the development of scenarios and models.

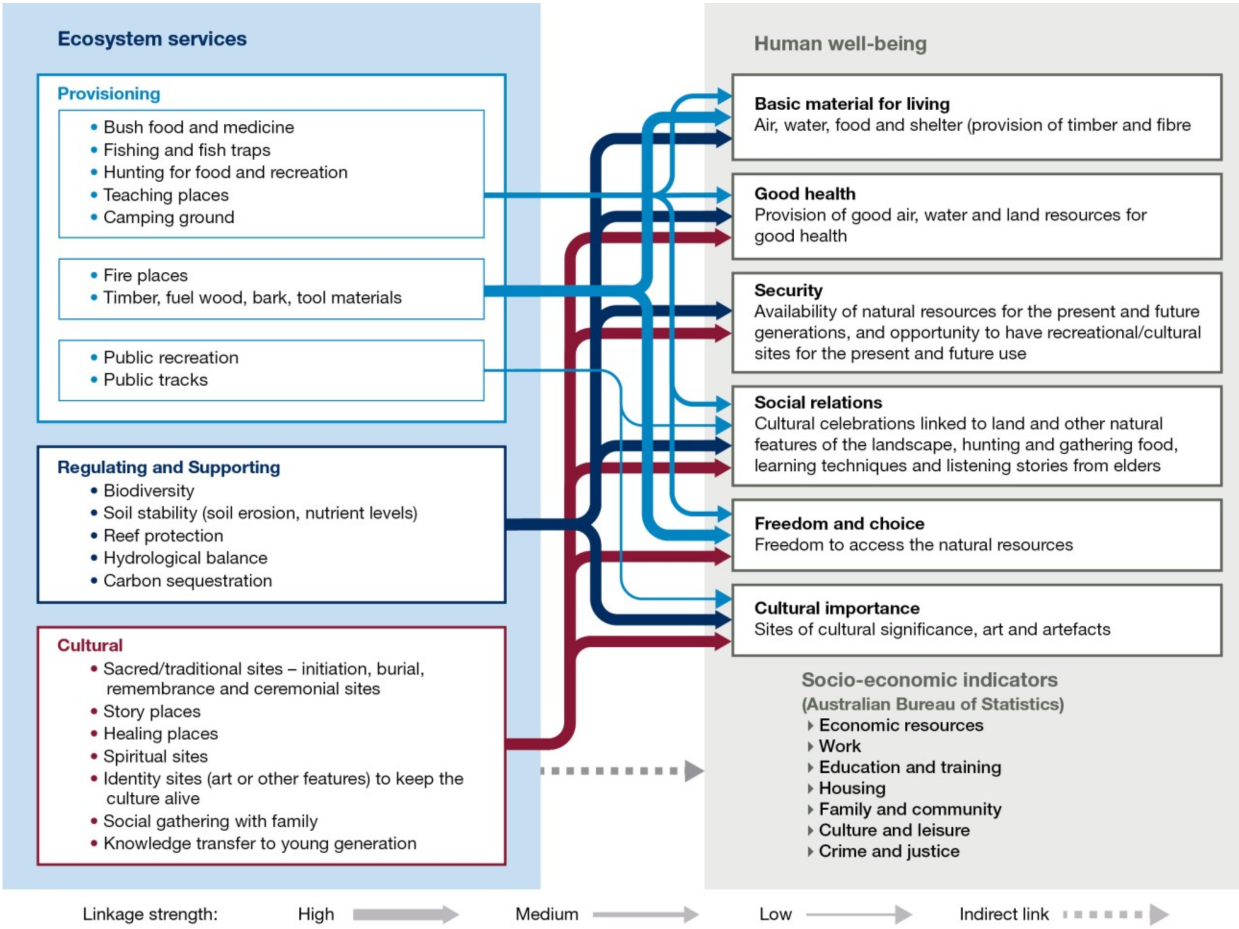



**Figure 7.6:** Relationships between ecosystem services and the constituents of well-being identified by the Mullunburra-Yidinji community, north Queensland (Modified from Sangha et al., 2011, David Publishing Company). Links between each ecosystem service and well-being are highlighted to demonstrate the importance of ecosystem services in terms of the well-being of indigenous peoples, and which indicators of well-being could be incorporated into the development of biodiversity and ecosystem services scenarios and models.

Indigenous knowledge of biodiversity and ecosystem services also links well with people's capabilities, which is important when discussing people's development from a policy decision-making perspective. For example, knowledge of bush food and medicine from local plants benefits people's health and enables them to develop a capability to pass on this knowledge to the next generation. As Sen (1999) suggested, enhancing people's capabilities (e.g. health, education) will enhance their well-being. This approach, which involves linking biodiversity and ecosystem services with indigenous capabilities, requires a new way of thinking about development, the economics of indigenous systems and related policies. The integration (co-perception) of knowledge from conventional and indigenous systems can help to consider the importance of biodiversity and ecosystem services in policy development decisions.

For example, Sangha et al. (2015) proposed an integrated well-being framework (Figure 7.7) focusing on the country – which is the indigenous perception of land systems in Australia – that equates to ecosystems. The framework links and equates various ecosystem services from the country (ecosystems) with the economic, cultural and social worlds of people. Such an integrated framework could be used as a tool for developing possible scenarios and models to suggest and analyse the role of biodiversity and ecosystem services in the economic and social worlds of indigenous and local communities.

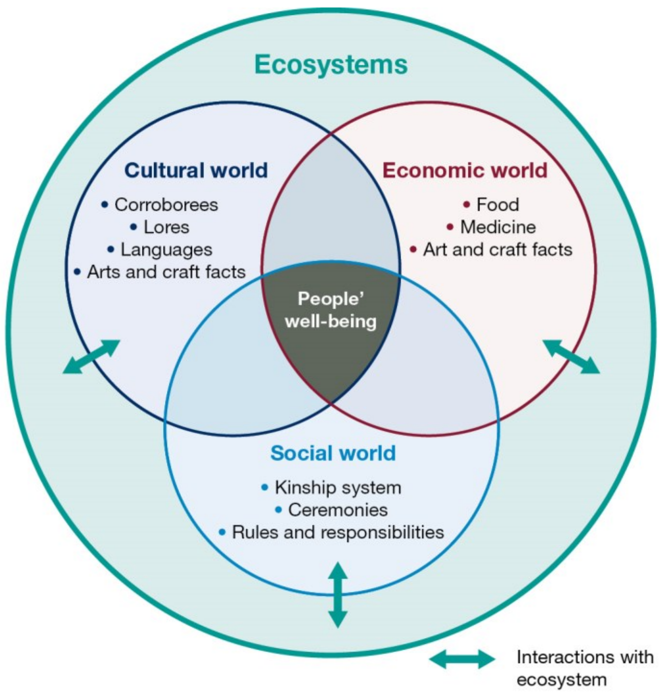



**Figure 7.7:** A proposed framework on how ecosystems (i.e. country in indigenous value system) deliver various ecosystem services (in the form of social, economic and cultural values) that are vital for indigenous well-being (Modified from Sangha et al., 2015, doi:10.1016/j.gecco.2015.09.001, https://creativecommons.org/licenses/by-nc/4.0/). In this framework, components of indigenous value systems are delivered through interactions between cultural, social and economic values and the natural world, and strong dependence of many aspects of indigenous well-being on the services provided by ecosystems, such as linkages with cultural rituals and ceremonies, traditional knowledge and governance systems, gathering of food and medicines, and indigenous arts.

To support the integration of traditional knowledge, people's capacities need to be identified through key stakeholders, as well as their interests and powers, and the feasibility of key stakeholders to participate in the development of relevant and inclusive biodiversity and ecosystem services scenarios and models (Table 7.2; CONDESAN and UMBROL, 2014). By incorporating traditional knowledge, biodiversity and ecosystem services scenarios and models can actually broaden the horizon and strengthen current knowledge systems.

# 7.5 Developing capacity for enhanced and meaningful multi-stakeholder engagement

Scenarios can prove to be useful tools for indigenous and local communities across the globe as they can encompass indigenous and local perspectives of natural systems from a much broader perspective than the biophysical or economic perspectives commonly used in many models. Combining scenarios with modelling can also be an effective tool for decision makers in terms of providing a long-term vision to support decisions. For example, each of the scenarios mentioned in Figure 7.4 could be further processed using InVEST or any other such model to project the outcomes over the long term.

## 7.5.1 Developing capacity for the effective engagement of stakeholders in assessments and other related activities at the national level

A wide range of qualitative and quantitative participatory methods are available to facilitate the engagement of stakeholders in scenario development.

These include, but are not restricted to: workshops; scenario-based stakeholder engagement; focus group meetings, questionnaire surveys, facilitated discussions and rankings; cooperative discourse; multi-criteria evaluation; conceptual system modelling; and dynamic systems modelling (Bousquet et al., 2002; Madlener et al., 2007; Magnuszewski et al., 2005; Kowalski et al., 2009; van den Belt, 2004; Castella et al., 2005; Renn, 2006; Tompkins et al., 2008 and others).

The key steps for facilitating effective stakeholder participation in scenario development (Reed et al., 2013) are:

1. Define the context (biophysical, socio-economic and political) and establish a basis for stakeholder engagement in scenario development;
2. Systematically identify and engage relevant stakeholders in the process;
3. Define clear objectives for scenario development, including spatial and temporal boundaries;
4. Select relevant participatory methods for scenario development.

The capacities required to involve stakeholders, the kinds of stakeholders and their levels of engagement are summarised in Table 7.8.

**Table 7.8:** Capacities required to engage with stakeholders and levels of involvement to integrate knowledge for scenario analysis and modelling.

| Level | Involved | Capacities |
|---|---|---|
| Local stakeholders and organisations | Local stakeholders, national and regional organisations | Representation<br>Leadership<br>Inclusion of biodiversity use/value into policy and decision making<br>Adaptability to ecosystem and functional change<br>Knowledge register<br>Information from peoples' systems of life<br>Feedback on indicators of direct drivers<br>Lessons learned<br>Integration of traditional and local knowledge<br>Procedures and legal instruments for biodiversity value and conservation |
| Institutions | Public and private regional associations | Transparency and credibility<br>Measurement of indicators of indirect and direct drivers<br>Interaction with local communities<br>Organisational support for biodiversity and ecosystem services<br>Transverse incorporation of biodiversity knowledge in the educational system<br>Generation of traditional indicators in the form of Ecosystem Data Groups<br>Exchange of information among Data Groups at regional level |
| Practice | Inter-scientific community; Associations that work with local stakeholders | Active participation<br>Measurement across qualitative and quantitative indicators in relation to the direct drivers<br>Technical integration and multilevel linkages<br>Recapitulation of lessons learned on managing and conservation<br>Inter-scientific dialogue<br>Establish and support networks on biodiversity and ecosystem services |

Learning occurs in both directions, with the enhanced understanding of local stakeholders in regional, national and international policy and management goals, as well as the incorporation of local knowledge into local, national and regional collaborative processes that support sustainable development and biocultural conservation. This requires collaborative dialogues between the stakeholders and decision makers (Rozzi et al., 2010). Educational initiatives are valuable outlets for enhancing partnerships between the scientific community and local communities through universities and school centres.

## 7.5.2 Developing capacity for the effective communication of the importance of biodiversity and ecosystems

Communication is crucial in disseminating the results of scenario and modelling exercises.

This requires clear communication towards target audiences through an appropriate means of communication. A lack of communication in real time could present a significant barrier to the effective participation of local communities, which influences the dialogue between communities and decision makers (Primack et al., 2001). The dissemination of knowledge is very important to enable local actors to take suitable decisions regarding management as part of the process of empowerment, and scientific research is more likely to be applied when there is an open dialogue between the different parties (Mauser et al., 2013).

Building confidence and trust in models and modelled outputs is a challenge as far as developing and using biodiversity and ecosystem services models and scenarios is concerned, and communication with local communities, stakeholders and industry can increase confidence in models and scenarios by enhancing the understanding of uncertainties in models. Complex dynamic models that simulate future ecosystem responses are often those which many have the least confidence in. The assessment and evaluation of model robustness and performance is therefore essential, as is estimating the effect of input datasets and uncertainty in model parameters on modelled outputs.

### 7.5.3 Developing capacity for the effective use of IPBES deliverables in the implementation of national obligations under biodiversity-related multilateral environmental agreements

To enable the communication of biodiversity and ecosystem services scenarios and models, they must be freely accessible and translated into products that are compatible with both local languages and scientific knowledge systems.

The co-dissemination of results may include publication of the acquired knowledge, also in an accessible language, and their translation into comprehensible and usable information for different stakeholders. This sharing of knowledge leads to open discussion and future research actions to target sustainability, which will then be jointly framed and initiate a new transdisciplinary research cycle (Mauser et al., 2013). A number of communication sources are available, such as graphical pamphlets, television and print media, educational systems, and internet and social media. The choice of communication media will depend on the community of interest and their technical capacity. The communication materials must contain key messages and have a presentation format that is relevant to the local communities (e.g. local language, drawings and printing, and characters) and must avoid excessive technical information. For example, if the aim is to register data on a species from a local perspective, the graphical material could link this with the needs of local people using agricultural calendars or cultural events. Highlighting the importance of a particular species in people's lives based on their current values and usages can also help in engaging and communicating with local people for future scenarios. An important part of information dissemination is that it should reach all sectors, including minorities, children, women and the elderly.

### 7.5.4 Developing capacity to strengthen networks and information sharing among different knowledge systems, including those of indigenous and local peoples

Long-term support for collaborative partnerships is important to ensure the long-term survival of traditional methods of managing common property resources and the integration of traditional knowledge into management decisions (Merino and Robson, 2006).

Global partnerships include organisations such as the Group on Earth Observations Biodiversity Observation Network (GEO BON) which coordinates activities relating to the Societal Benefit Area on Biodiversity of the Global Earth Observation System of Systems (GEOSS), and the UNESCO Sacred Natural Sites programme. Similarly, the UN Environment Programme (UNEP) Sub-Global Assessment network strengthens regional and global networks among scientists. Most existing networks are for model practitioners, scientists and policymakers working on the development and implementation of models and scenarios. However, there are few networks for local and indigenous communities. Similar networks need to be supported for indigenous and local communities at the local, regional and global scales, and IPBES can stimulate such a platform.

## 7.6 Consolidation, strategy and recommendations

Based on the capacity-building requirements identified for biodiversity and ecosystem services models and scenario analysis, the following broad recommendations are proposed to improve the use and application of biodiversity and ecosystem services models and scenario analysis:

### 7.6.1 Close capacity gaps for regional biodiversity and ecosystem services scenarios and models

IPBES could:

- Produce manuals and guidelines to improve common data users' understanding, possible methodologies, and the limitations of biodiversity and ecosystem services scenarios and models, adapted to the situation and capacities of the different UN regions.
- Develop brochures and booklets about biodiversity and ecosystem services scenarios and models that are adapted to the different user groups and, as such, enable them to tailor and package their scenarios and models in ways that are more useful for decision makers.
- Establish international forums for biodiversity and ecosystem services scenarios and models managed by highly qualified experts that have the necessary skills to translate scientific concepts into concepts that users understand and can use, without distorting the concepts. These forums could serve as tools for people to ask questions and interact with other users of models and scenarios, and to promote knowledge exchange and capacity development within and between regions. The experts who manage these forums should be chosen to represent the different UN regions and should have an in-depth understanding of users' needs and potential opportunities for developing biodiversity and ecosystem services scenarios and models in their regions.
- Use the lessons learned from previous global and regional assessments to define the further critical skills and expertise required to effectively develop more integrated biodiversity and ecosystem services scenarios and models to support decision makers.

### 7.6.2 Develop capacity for effective participation in IPBES assessments

IPBES could:

- Develop global, regional and national lists of open source and freely accessible software and tools (e.g. Deliverable 3d) that will support the development of successful biodiversity and ecosystem services scenarios and models. All tools (models, software and databases) should be well documented, in an intelligible language that the users can understand. Metadata associated with models should be written following international standards, fully illustrated and intelligible to both specialists and non-specialists.
- Run and maintain regular in-person and/or online courses at global and regional/national scales, providing training on the use and applications of different models and software tools.
- Use and build upon the upcoming global, regional and sub-regional assessments to establish networks of mentoring schemes for early career scientists and researchers. This will seek to

facilitate the establishment of mentoring relationships between early-career scientists and researchers working in the field of ecosystem assessments/services or established assessment practitioners, to promote capacity development for undertaking and using current or upcoming ecosystem assessments.

- Develop global and regional 'fellows programmes' on integrated biodiversity and ecosystem services scenarios and models for young scientists, to transfer the gained experience to the national levels.
- Build partnerships between the IPBES Task Force on Capacity Building and other global programmes and initiatives to provide a wide range of training courses, workshops, internships and collaborative projects with universities in developing and developed countries to train practitioners on tools and software for scenario development and modelling.
- Provide funds, in cooperation with other international and regional donors, to strengthen national institutions and infrastructure on biodiversity modelling and scenario usage through multidisciplinary research, activities, planning and budgeting.

### 7.6.3 Promote dialogue between different world views and knowledge systems

IPBES could:

- Encourage participation in and contributions to the existing global scenarios, models and database infrastructure to enhance their capacities instead of building new infrastructure, thus minimising the duplication of efforts.
- Initiate the development of a free Android-based and/or iOS-based application about IPBES and the biodiversity and ecosystem services models and scenario analyses presented in Deliverable 3c to take advantage of new technologies to reach different stakeholder groups.
- Encourage IPBES to effectively engage local and indigenous knowledge from the first stage of planning scenarios in order to allow co-definition of the problem, to increase trust and understanding between participatory stakeholders, and to reduce uncertainty in the scenarios.
- Develop an integrated set of biodiversity and ecosystem services indicators based on scientific and traditional knowledge. These indicators could include or have links to local and regional traditional knowledge systems to enhance socio-ecological scenarios and models.
- Produce standardised training modules that are made available to government officials, decision makers and practitioners as a means of strengthening their capacity to draw appropriately on available data. The training modules could also raise awareness of the available data as it evolves.

### 7.6.4 Improve capacity building relating to data management and infrastructure

IPBES could:

- Invite IPBES countries to participate in matchmaking projects, programmes and events to enhance resource sharing for biodiversity and ecosystem services modelling and scenario development. Data sharing can demonstrate the cost effectiveness of forecast-based policymaking in resource management sectors such as agriculture and biotechnology, protected area management, forestry, nature conservation and coastal zone management.

- Develop tools to improve and adapt models of species occurrence data.
- In cooperation with existing regional and international institutional networks and respective human resources, cultivate cooperative web-based digital products on biodiversity, modelling, scenario building, accuracy improvement and implementation that are needed to support robust modelling outcomes.
- Initiate the development of a set of biodiversity and ecosystem services indicators and indices, whereby indirect statistics of existing biodiversity indicators/indices and platforms could serve as a starting point.

### 7.6.5 Incorporate traditional and local knowledge

To achieve the effective integration of traditional knowledge and socio-ecological feedback into models and scenarios for biodiversity and ecosystem services, IPBES could:

- Work to identify universities, research institutions and NGOs with experience and/or existing relationships that enable the integration of traditional and local knowledge into the development of biodiversity and ecosystem services scenarios or models over both short and long timescales.
- Build the capacity of ILK networks by identifying leadership and educational opportunities and mechanisms to enhance communication between indigenous organisations and local governments.
- Establish agreements of cooperation between local governments and indigenous technical personnel and organisations for knowledge transfer and for coordination with educational entities to promote the incorporation of information on biodiversity and ecosystem services into the educational curriculum.
- Develop policy-relevant scenarios backed by rigorous scientific data and local knowledge for decision makers. These should properly integrate scientific, social, economic and local information to tell a good storyline. Apply a balanced approach (just enough data to appropriately inform the stakeholders) to develop scenarios and provide sufficient scientific data to help policymakers comprehend the impacts or changes under a given scenario.

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# 8 Improving the rigour and usefulness of scenarios and models through ongoing evaluation and refinement

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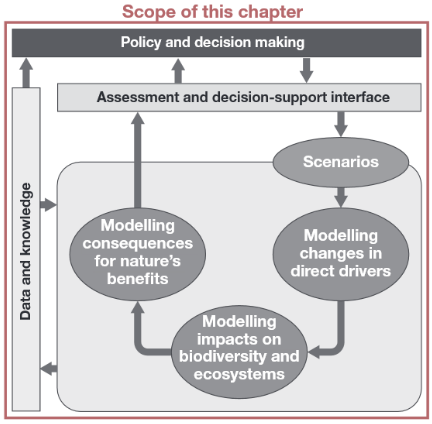
**Purpose of this chapter:** Adopts a more forward-looking perspective than the previous seven chapters; and thereby identifies major directions, both in underpinning science and in practical application, that need to be pursued to ensure the future rigour and utility of scenarios and models of biodiversity and ecosystem services.

**Target audience:** While less technical than most of the preceding chapters, this chapter is targeted mainly at readers seeking guidance on where best to direct future effort and support in developing and applying scenarios and models.




## Key findings

**There are significant gaps in data availability and data access for biodiversity and ecosystem services.** The spatial, temporal and taxonomic coverage and resolution of monitoring of biodiversity change is heterogeneous. There are also gaps in information on social demand for ecosystem services and in high-resolution data of ecosystem properties relevant for ecosystem services. Much progress has been made in mobilising data on biodiversity and ecosystem services, but significant barriers remain to data sharing. More efforts are required to provide easier access to well-documented data and models (Sections 8.2.2 and 8.2.3).

**There are already many models available to assess the impacts of drivers on biodiversity change and ecosystem services; however, important gaps remain**. These include gaps on (i) linkages between biodiversity and ecosystem services; (ii) ecological processes at temporal and spatial scales relevant to the Intergovernmental Platform on Biodiversity and Ecosystem Services (IPBES) assessments, including species interactions and community dynamics; (iii) early warning systems to anticipate ecological breakpoints and regime shifts; and (iv) coupling of, and feedbacks between, social and ecological components of ecosystems (Section 8.3.1).

**Scenarios can allow the effective use of data and models in decision making.** Both short-term scenarios (10 years) examining alternative policy options and long-term scenarios examining plausible futures are useful in assessing the impacts of drivers on biodiversity and ecosystem services. Exploratory scenarios foster creative thinking and the exchange of viewpoints between different stakeholders, but do not always provide clear actions that decision makers can implement to reach desirable outcomes. Normative scenarios are more likely to provide clear policy pathways but have been criticised for being value-laden (Section 8.4).

**Scenarios can be improved through an iterative process that includes the steps of: engaging stakeholders, linking models to policy options, managing uncertainty, communicating the results and bringing scenario outcomes to policymaking.** It is critical that assessments identify stakeholders relevant at the scale of the problem, including scientists, decision makers and people with indigenous and local knowledge (ILK), and engage them early on in the modelling and scenario analysis process (Section 8.4.1). Models and scenarios can improve the transparency of policymaking, by rendering the assumptions explicit and facilitating the comparison of multiple options (Section 8.4.2).

# Key recommendations

**IPBES could engage with existing processes on increasing data collection and data sharing.** Key tasks are to identify common metrics for monitoring, modelling and reporting biodiversity and ecosystem services and to develop cost-effective approaches that are geared towards the needs of users at multiple scales (Sections 8.2.1 and 8.2.2). The Task Force on Knowledge, Information and Data (Deliverable 1d) could adopt existing data and model documentation standards and expand those as needed, make use of existing central repositories, liaise with relevant organisations to develop new ones, and participate in ongoing efforts to assure proper credit to data and model providers.

**The IPBES Expert Group on Scenarios (Deliverable 3c) is encouraged to develop guidelines for the verification and validation of models, and for assessing and managing uncertainty in scenario analysis and modelling**. These guidelines need to be regularly updated based on scientific developments (Section 8.3.2). Complementary to visual validation, statistical analyses and accuracy tests are pivotal to make model validation and model comparisons robust, general and quantitative. It is important for the IPBES regional and global assessments to use verified and validated models with a relevant pedigree and to adopt appropriate methods for incorporating and communicating uncertainties. Depending on the context and topical relevance, multiple models of differing complexities and types could be used to address structural uncertainties.

**Thematic, regional and global assessments are encouraged to use both short-term (e.g. 10 years) and long-term scenarios (e.g. 50 years) to assess the future of biodiversity change and ecosystem services and their implications for human well-being**. For the regional assessments, existing long-term scenarios from other initiatives can be adopted and downscaled to the regions. For the global assessment, a new set of long-term exploratory scenarios could be developed around key issues specific to biodiversity and ecosystem services (including those related to Sustainable Development Goals (SDGs)), as identified by the relevant stakeholder community. Short-term scenarios comparing policy options using models and qualitative information can be developed both in regional and global assessments (Section 8.4.2).

**The Task Force on Capacity Building (Deliverable 1a/b) could support the use of models and scenarios in assessments at different scales, as well as interaction among social and natural scientists and multiple stakeholders**. This includes activities that give planners and policymakers a better understanding of models and scenarios, including limitations and uncertainties, and activities that assist modellers in engaging further with policy and planning processes. Further research is needed on developing robust methods to elicit ILK for the development of models and scenarios (Sections 8.4.2, 8.4.3 and 8.4.4). The Task Force on Indigenous and Local Knowledge (Deliverable 1c) may liaise with the Task Force on Capacity Building to foster this research.

**The follow-up work to the assessment on scenarios and modelling (Deliverable 3c), conceptualisation of values (Deliverable 3d) and policy support (Deliverable 4c) could ensure that the review of available policy-support tools and methodologies for scenario analysis and modelling**

**continues to reflect best available science.** Because of ongoing research in and the rapid progress being made on many aspects of scenario analysis and biodiversity and ecosystem services modelling, there is a need to continually update the review of available policy-support tools and methodologies for scenario analysis and the modelling of biodiversity and ecosystem services. Furthermore, the Task Force on Knowledge, Information and Data (Deliverable 1d) could develop a process of prioritisation of research needs, to encourage basic research that advances scenario analysis and modelling in contexts and at scales that are relevant to IPBES with the ultimate objective of decision support. This especially concerns research on including socio-cultural aspects in modelling and scenario development (Section 8.3.1.3).

## 8.1 Introduction

Previous chapters demonstrated the variety of approaches to scenario analysis and modelling that can be used to inform decisions and evaluate policy options. Scenario analysis and modelling can address issues ranging from the local scale, such as assessing consequences of municipal land-planning options for ecosystem services and biodiversity, to the global scale, such as the impacts of alternative pathways of population economic growth on biodiversity and ecosystem services. Although IPBES assessments range only from sub-regional to global scales, this chapter also provides information relevant for local scales. Previous chapters identified the problems or challenges, and reviewed existing solutions, for the use of models and scenarios in assessments of biodiversity and ecosystem services. The goal of this chapter is to chart the way forward for additional research and development that is required to take the use of models and scenarios to a whole new level of rigour and utility.

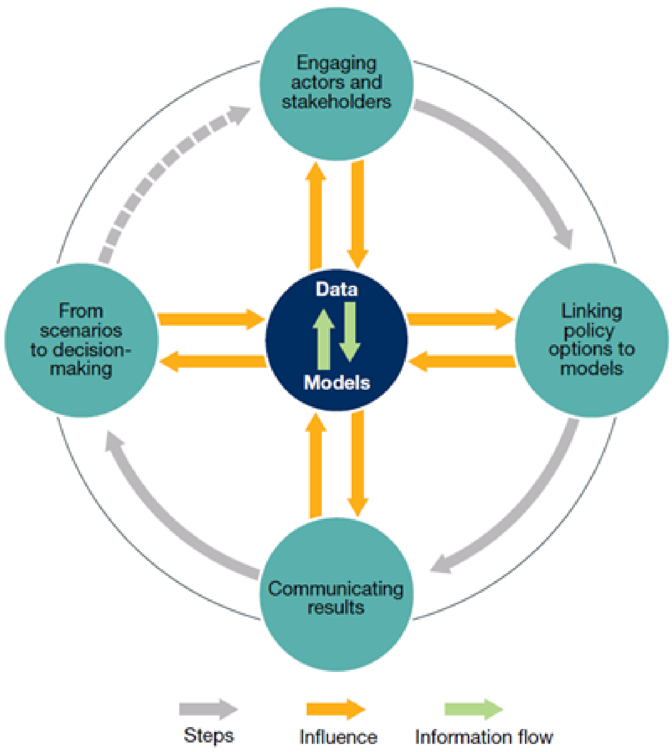



**Figure 8.1:** Scenario development and analysis process involving steps (in white ovals) such as engaging actors

and stakeholders (including ILK), with each step interacting with the data and models (green arrows) and with information flow between models and data (red arrows). The dashed arrow indicates that the policy assessment involves several instances and actions, repeatedly involving actors and stakeholders; hence the iterative nature of this process.

This chapter is organised into three main sections. We first discuss approaches to improving the data used to calibrate and validate biodiversity and ecosystem services models, emphasising linkages to various existing initiatives for biodiversity monitoring at national, regional and global scales. We then discuss basic and applied science research needed to improve models of biodiversity and ecosystem services, both by promoting the development of new models and by encouraging and facilitating functional linkages among existing models and modelling platforms. Finally, we discuss directions for improving the relevance of scenarios for policymaking. We consider four key steps of the iterative cycle of scenario development that are supported in models and data (Figure 8.1): (1) engaging actors and stakeholders, (2) linking policy options to models and scenarios, (3) communicating results, and (4) using the scenario results and analysis for decision making. In our discussions we take 'ecosystem services' to be synonymous with 'nature's benefits to people', following the IPBES Conceptual Framework ({Díaz, 2015 #1470}).

This chapter emphasises quantitative approaches to measure and forecast biodiversity and ecosystem services. However, we also cover interdisciplinary and transdisciplinary approaches, involving the social sciences and stakeholders, and point out corresponding research needs and best practices.

## 8.2 Improving data

### 8.2.1 Identifying common metrics

Biodiversity has multiple dimensions, including genetic diversity, species diversity, functional diversity and ecosystem diversity, and can be measured in a multitude of ways (Noss, 1990; Pereira et al., 2012). Similarly, there are many ecosystem services and each ecosystem service can be quantified using different approaches, including biophysical, cultural and economic measurements (Daily et al., 2009; Hauck et al., 2016). Important challenges remain in bridging towards the socio-cultural values of ecosystem services (Martín-López et al., 2012). The values of nature, nature's benefits to people and good quality of life are plural and can be considered from diverse dimensions, some quantifiable and others not. Researchers often face the challenge of accessing adequate data for the calibration and validation of models, as different initiatives monitor differently and even have diverging epistemologies. There is a lack of harmonisation and integration of monitoring methods, datasets and approaches across observation communities (e.g. different research communities, governmental agencies, non-governmental organisations) and across countries (Pereira et al., 2013).

A key challenge is to identify common metrics that could be used by the modelling and observation communities. A common set of metrics for the observation and modelling of biodiversity and ecosystem services would foster collaboration between the modelling and observation communities. This would promote the integration of data from different sources, foster the development of approaches to fill data gaps, and facilitate the calibration and validation of models and scenarios and inter-model comparison.

Two complementary approaches, at different levels of data abstraction, currently show promise (Table 8.1): the Essential Biodiversity Variables being promoted by the Group on Earth Observations

Biodiversity Observation Network (GEO BON, www.geobon.org) (Pereira et al., 2013), and the biodiversity indicators adopted by the Convention on Biological Diversity (CBD) and supported by the Biodiversity Indicators Partnership (www.bipindicators.net) to assess progress towards the 2010 target and the 2020 Aichi biodiversity targets (Butchart et al., 2010; sCBD, 2010; Tittensor et al., 2014; sCBD, 2014).

In recent years, scientific communities of different physical and biological phenomena have started to identify essential variables that are critical for monitoring and modelling. The first such effort was the identification of the Climate Essential Variables by the Global Climate Observing System. Similarly, GEO BON has developed a process to identify Essential Biodiversity Variables. The idea behind this concept is to identify, using a systems approach, the key variables that should be monitored to measure biodiversity change. The Essential Biodiversity Variables are an intermediate layer of abstraction between the raw data from in situ and remote sensing observations and the derived high-level indicators used to communicate the state and trends of biodiversity. These variables can be used as the main system variables in models of the whole biosphere or parts of it, and can then be used to compare model simulations with data. For example, the population abundance variable is defined as a three dimensional matrix of population abundances per species, per location, per time. A gridded dataset of population abundance for a group of species requires the integration of population estimates from different methods and observers, and the interpolation of gap areas with models. Models for interpolation can use as inputs climate variables and other environmental variables, including variables that can be remotely sensed. A list of 22 Essential Biodiversity Variable candidates has been identified and organised into 6 major classes (Pereira et al., 2013): genetic composition, species populations, species traits, community composition, ecosystem structure and ecosystem function (Table 8.1). Efforts are ongoing to identify appropriate monitoring schemes, propose data standards and develop global or regional datasets for each variable.

Some Essential Biodiversity Variables measure directly the supply of ecosystem services such as nutrient retention or net primary productivity (a measure closely related to the carbon sequestration service). Essential Biodiversity Variables can also be used to measure the supply of services dependent on the distribution of particular species, such as wild animals used for food or medicine (Díaz et al., 2015). However, for some of these and other nature's benefits, it is important to look at the entire ecosystem service supply chain, and incorporate the role of human activities and social preferences in models (Tallis et al., 2012; Karp et al., 2015).

**Table 8.1:** Examples of common metrics for observation, reporting and modelling for each class of Essential Biodiversity Variables. Some Essential Biodiversity Variables have related indicators that are used to assess progress towards the CBD 2020 targets. Essential Biodiversity Variables development focuses on how to monitor or model, while indicator development focuses on how to report or communicate. Examples of models that project the evolution of an Essential Biodiversity Variable metric or an Aichi indicator under different scenarios are also provided. References: [1]Pereira et al., 2013; [2]CBD, 2015; [3]Brook et al., 2000; [4]Christensen and Walters, 2004; [5]Harfoot et al., 2014; [6]Guisan and Thuiller, 2005; [7]Visconti et al., 2015; [8]Jetz et al., 2012 [9]Newbold et al., 2015; [10]Alkemade et al., 2009; [11]Hurtt et al., 2011; [12]Nemec and Raudsepp-Hearne, 2013; [13]Sitch et al., 2008.

| Essential Biodiversity Variable classes[1] | Essential Biodiversity Variable metrics[1] | Aichi indicators[2] | Models |
|---|---|---|---|
| Genetic composition | Number of animals of each livestock breed and farmed area under each crop | Trends in genetic diversity of cultivated plants and of farmed animals | - |
| Species populations | Population abundance of selected species or functional groups | Trends in species populations; Trends in proportion of fish stocks outside safe biological limits; Trends in species extinction risk | Population viability analysis[3]; Trophic models of ecosystems[4,5] |
| | Species distribution of selected species | Trends in species extinction risk | Species distribution models[6]; Habitat suitability models[7,8] |
| Species traits | Leaf senescence for selected species | - | - |
| Community composition | Species richness of a community | Trends in degradation of forest and other habitats | Dose-response models[9,10] |
| Ecosystem structure | Proportion of cover of each habitat type | Trends in extent of forest and other habitats | Integrated assessment models[11] |
| Ecosystem function | Nutrient retention | Trends in nutrient levels | Ecosystem service models[12] |
| | Net primary productivity | Trends in carbon stocks | Dynamic global vegetation models[13] |

For instance, supply for wood production can be assessed by standing biomass, demand by timber harvest, and benefit by the market value of timber products (Tallis et al., 2012). GEO BON has proposed a set of metrics to monitor ecosystem services globally at different stages of the supply chain (Tallis et al., 2012; Karp et al., 2015). The Mapping and Assessment of Ecosystem Services initiative has identified a wide range of indicators and measures for provisioning, regulating and cultural services tailored to each major category of ecosystem in Europe: forests, agro-ecosystems, freshwater and marine (EC, 2014). A set of ecosystem service measures has also been proposed by the Final Ecosystem Goods and Services Classification System (FEGS-CS) (Landers and Nahlik, 2013) and by the Experimental Ecosystem Accounting of the United Nations System of Environmental Economic Accounting (UN SEEA) (UN et al., 2014). Many ecosystem services (e.g. some regulating services) cannot be easily directly observed and models play a key role in their assessment (Table 8.1).

The identification of common metrics can also be based on aggregated indicators and indices (van Strien et al., 2012). Over the last decade, several biodiversity indicators have been used to report on biodiversity change at the national and global levels (Butchart et al., 2010; Tittensor et al., 2014; CBD, 2015). The Biodiversity Indicators Partnership (http://www.bipindicators.net) has played an important role in this process. Indicators condense a wealth of data into a few values. For instance, one specific indicator for trends in species populations (Table 8.1), the Living Planet Index (LPI), condenses information on population counts of several thousands of vertebrate populations into a single global value per year, which informs on global vertebrate population reductions relative to a base year. Another specific indicator for trends in species extinction risk (Table 8.1), the Red List Index (RLI) (Butchart et al., 2004; Baillie et al., 2008), condenses assessments of species status of >20,000 species into a single value for a time point, which can be compared with values from previous time points to assess whether there has been an acceleration or deceleration in biodiversity loss.

It is possible to model either the more disaggregated data of each Essential Biodiversity Variable or the more aggregated data of biodiversity indicators and indices. For instance, many models are available to develop scenarios for population abundances or occupancy across ranges of individual species or groups of species (Table 8.1). However, it is also possible to model the dynamics of aggregated indices such as mean species abundance or species richness at local to global scales (Nicholson et al., 2012). A particular challenge of using species richness or species abundance indices rather than the disaggregated data is the choice of appropriate aggregated metric. A wide range of metrics is used to describe change in community composition, such as species richness, phylogenetic diversity, Simpson's diversity index, geometric mean abundance and arithmetic mean abundance, just to name a few (van Strien et al., 2012; Buckland et al., 2005; Lyashevska and Farnsworth, 2012). It is also possible to focus on a subset of species, such as rare or endemic species versus abundant species, or threatened versus non-threatened species. The Essential Biodiversity Variables framework is particularly flexible in this regard, as calculating an index of an Essential Biodiversity Variable can result in another Essential Biodiversity Variable: for example using occupancy data for a set of species in a community to calculate species richness (Table 8.1). Furthermore, Essential Biodiversity Variables can be modelled globally, integrating in situ observations and remote sensing, and used as inputs to the calculation of spatially explicit indicators (GEO BON, 2015).

Understanding the upstream drivers and pressures and the downstream impacts and management responses are crucial in assessing biodiversity and ecosystem services. The drivers-pressures-states-impacts/benefits-responses (DPSIR) indicator framework allows for the consistent assessment of the dynamics of social-ecological systems (Sparks et al., 2011), and it is used to develop scenarios for biodiversity and ecosystem services (Pereira et al., 2010). The CBD Aichi biodiversity targets for the year 2020 are organised into five strategic goals that closely follow the DPSIR framework and can be assessed by using indicators for each target component (Tittensor et al., 2014; Leadley et al., 2014b). The DPSIR framework also makes clear that the variables used as outputs of some models can be the inputs of other models. For example, a socio-economic model may project changes in the harvest pressure of fish stocks, leading to changes in the abundance of different species. In turn, this change in ecosystem state may lead to changes in fish provisioning from the ecosystem. Therefore, the choice of metrics has to take into account the interoperability of different models. Finally, metrics or indicators can be chosen so that they are able to detect biodiversity trends reflecting changes in pressures or policy and management (Nicholson et al., 2012). Indicators at regional scales or for specific groups of taxa (e.g. taxa vulnerable to a specific driver) may be more likely to do so than generic global indicators.

It is important for IPBES to engage in processes that aim to identify common metrics of biodiversity and ecosystem services, to guarantee that the metrics, associated monitoring methods and data standards serve the needs of assessment users. Therefore, the participation of all IPBES stakeholders is important to ensure a balanced choice of metrics.

Regional and global IPBES assessments could report results of models and scenarios using a set of common metrics for biodiversity, including selected Essential Biodiversity Variables and/or Aichi indicators (Table 8.1). Models of nature's benefits could use the standard classification of ecosystem services, such as the Common International Classification of Ecosystem Services (CICES) (EC, 2014), and common metrics such as the ones identified by GEO BON (Tallis et al., 2012) or the UN SEEA (UN et al., 2014). Indeed, the Task Force on Data and Knowledge has already proposed a list of indicators that could be used by regional IPBES assessments (IPBES/3/INF/4, http://ipbes.net). This set of indicators could be further explored by the Task Force on Modelling and Scenarios, which could also update regularly the guidelines presented in the current report.

Regarding socio-cultural values, it is important to recognise the complexity of incommensurable values and, if possible, find practical ways to deal with this (IPBES/3/INF/7, http://ipbes.net/). Assessments could be explicit about which value dimensions were included in the scenarios, which could not be included, and what the implications of this selection are (IPBES/3/INF/4, http://ipbes.net/).

### 8.2.2 Increasing data availability for model calibration and validation

Despite recent increases in the variety and amount of biodiversity-related data, there are significant gaps with respect to quantity and quality (Brooks and Kennedy, 2004) and significant biases in the availability of biodiversity and ecosystem services data (Box 8.1). Reasons for these gaps include lack of financial support for long-term monitoring, lack of local capacity, and limited international collaboration on developing globally representative monitoring programmes (Scholes et al., 2012).

Different technical and strategic approaches could be taken to overcome biases and gaps in data availability for biodiversity and ecosystem services. IPBES could identify critical gaps and promote the enhancement of (i) monitoring programmes, (ii) the mobilisation of data, and (iii) modelling for interpolation, estimation and other methods for filling data gaps.

In many cases, existing databases can be improved with concerted and coordinated efforts to increase spatial (regional) coverage, spatial resolution (e.g. smaller grid size or denser sampling points), temporal resolution (regular and frequent observations), and temporal coverage (long-term, sustainable monitoring for the future; historical reconstruction for the past). For example, the Projecting Responses of Ecological Diversity In Changing Terrestrial Systems (PREDICTS) database collected data from the existing literature relating to 78 countries representing over 28,000 species (Hudson et al., 2014), including invertebrates, vertebrates and plants in terrestrial ecoregions around the world. However, the areas covered by the database are not balanced, but representative of the data availability (see also Figure 8.2), indicating a need for improvements in existing data. Monitoring programmes could implement a data strategy that supports intelligent choices about what and how to measure (Section 8.2.1) and be cost-efficient, sustainable through space and time, and effective, avoiding duplication (Box 8.2). For instance, in terms of taxonomic coverage, adding large numbers of species in poorly studied taxonomic groups may not be cost effective. However, a taxonomically sampled approach, as used in the Sampled Red List Index (Baillie et al., 2008), can provide taxonomic coverage in a cost-effective way. It would also be beneficial if monitoring programmes were to expand their efforts in observations of the ecosystem services of most importance to human well-being, and if the data were more accessible (see Section 8.2.3).

New and promising approaches to obtaining data and building and curating datasets include citizen science and crowd-sourcing (Silvertown, 2009, Wiggins and Crowston, 2011), as well as new technological tools such as automated data collectors and sensor networks that are embedded in the environment (Collins et al., 2006; Porter et al., 2009; Rundel et al., 2009; Benson et al., 2009). The new field of eco-informatics envisions building ecological datasets in the context of a 'data life cycle' that encompasses all facets from data generation to knowledge creation, including planning, collecting and organising data, quality assurance and quality control, metadata creation, preservation, discovery, integration, and analysis and visualisation (Michener and Jones, 2012). Eco-informatics tools that support and assist various steps of the data life cycle include data management planning tools (e.g. http://dmp.cdlib.org/); metadata standards and tools; relational databases that allow the specification of constraints on the types of data that can be entered (i.e. data typing), assuring data integrity; scientific workflow systems such as Kepler, Taverna, VisTrails and Pegasus (see Section 8.3.1.2); and cloud-computing resources.

In some cases, gaps in datasets can be filled using quantitative approaches such as statistical and modelling methods. One approach is imputation, which is often used when analysing large datasets of demographic traits (e.g. Di Marco et al., 2012; Penone et al., 2014), but this relies on the assumption that relationships that exist in the data are also valid for the missing data. Another option for filling data gaps is to make inferences based on allometric relationships between biological variables such as body size, metabolic rates, population density, generation time and maximum population growth rate (e.g. Damuth, 1987). Although allometric relationships have been used, for example, in size-structured food web models (Blanchard et al., 2009) and in models of energy budgets (Simoy et al., 2013), large uncertainties in the predicted values limits their usefulness in estimating parameters of predictive dynamic models at the species level. However, they may be useful, even in this context, if limited to groups of functionally related species (such as herbivorous mammals). A third approach involves sampling demographic parameters of population models using a 'generic life history modelling' approach. Although linking ecological niche and population models gives more realistic predictions of the effects of changing environmental conditions on species (Keith et al., 2008), the widespread application of such coupled niche-population models is hampered by the availability of species-specific demographic data. Generic life history modelling (Pearson et al., 2014; Stanton et al., 2015) gets around this problem by using ensembles of population models designed to encompass the full set of life history parameters characteristic of a particular group of species. This approach avoids the need to obtain species-specific demographic parameters, which are rarely known, and enables the generalisation of results beyond the well-studied species; however, this is achieved at the cost of not being able to make species-specific predictions of population dynamics (Pearson et al., 2014).

Remote sensing and in situ data are vital for modelling and monitoring environmental parameters relevant for biodiversity conservation (Buchanan et al., 2009; Kogan et al., 2011; Skidmore et al., 2015). Satellite remote sensing is useful for collecting data across different spatial and temporal scales. However, many users still lack the capability to deal with these data. Access to training and education in using satellite-based observations will be essential in the future to address this issue (Turner et al., 2015). Some initiatives for increasing access to remote sensing data globally are the GEO (www.earthobservations.org), the European Space Agency's Climate Change Initiative (Bontemps et al., 2011), the EU Copernicus Programme, and the Committee on Earth Observation Satellites (ceos.org).

**Box 8.1:** Biases and gaps in data availability of biodiversity and ecosystem services

- **Regional biases in coverage:** Historically, ecologists have studied non-urban but relatively accessible areas in wealthy countries, resulting in a very uneven global distribution of study areas (Figure 8.2). The disparity among terrestrial, freshwater and marine realms is also noteworthy (Loh et al., 2005).
- **Taxonomic biases in coverage:** Ecological studies have focused disproportionately on conspicuous species. Vertebrates, particularly birds and mammals, are much more often the focus of ecological studies than invertebrates and plants (Pereira et al., 2012). One of the most popular indices for measuring global biodiversity change, the Living Planet Index (LPI), is based on vertebrate populations only (Loh et al., 2005).

- **Spatial and temporal resolution:** Most ecological studies either have a high spatial resolution and small spatial extent, focusing in detail on small areas, or have a low spatial resolution and focus on larger regions. For some scenario analysis and modelling approaches, high resolution data with global coverage are needed (Pereira et al., 2010). Such data exist for some biodiversity-related variables (such as forest cover data available at http://earthenginepartners.appspot.com), but this is rare.
- **Thematic gaps:** There is a lack of regional and global consensus on what to monitor. Some Essential Biodiversity Variable classes such as species traits and genetic composition have received less attention from monitoring programmes than others such as species populations. Regulating and cultural ecosystem services and particularly their benefits for populations are not monitored or only partially monitored in most places (Tallis et al., 2012).

**Box 8.2:** Data strategy (modified from Scholes et al. (2012) and other sources). Required properties of IPBES-relevant data:

1. Data that are aligned with the needs of scenario analysis and modelling at global, regional and local scales are relevant and useful for decision making.
2. Global in coverage, but with sufficient resolution and accuracy at subnational scales to be useful to the main decision makers at this scale.
3. Statistically sound basis for repeated measurements of biodiversity.
4. Following best practices for metadata specification.
5. Provisions for coordinating and managing data that are collected by disparate institutions and individuals for different purposes.
6. Sufficiently comprehensive in terms of taxonomic coverage.
7. Quality controlled, with well-defined standards for formats, codes, measurement units and metadata; traceability of the observation (including place and time of origin, the techniques used to make the observation, and methods used to modify the data); enforced data typing.
8. Cost efficient. Avoiding duplicate work in recording or analysing the same observations for the same time period.
9. Sustained. Ensuring data continuity and comparability over time, including provisions for long-term storage and data management.
10. Adaptive. Responsive to new technical possibilities, emerging societal needs and changing system states.
11. Interoperable. Data available to (and discoverable by) other parts of the system, with tools to enable the analysis of data from different parts together. Requires metadata (see above) and the harmonisation of observations, analysis and data exchange standards and protocols.

Metrics and indicators of the quantity and quality of ecosystem services are essential for knowing if these services are being sustained or lost or how they need to be managed in order to sustain human well-being and biodiversity (Layke et al., 2012). While some ecosystem services (e.g. providing goods) can be directly quantified, most regulating, supporting and cultural services are less straightforward to quantify, requiring indicators or proxy data (Egoh et al., 2012). The development of robust indicators is an important step towards mapping ecosystem services and meeting biodiversity targets (Egoh et al., 2012). In recent years, ecosystem services modelling has improved with governmental demand for standardised practices to measure, value and map ecosystem services (Waage and Kester, 2014).

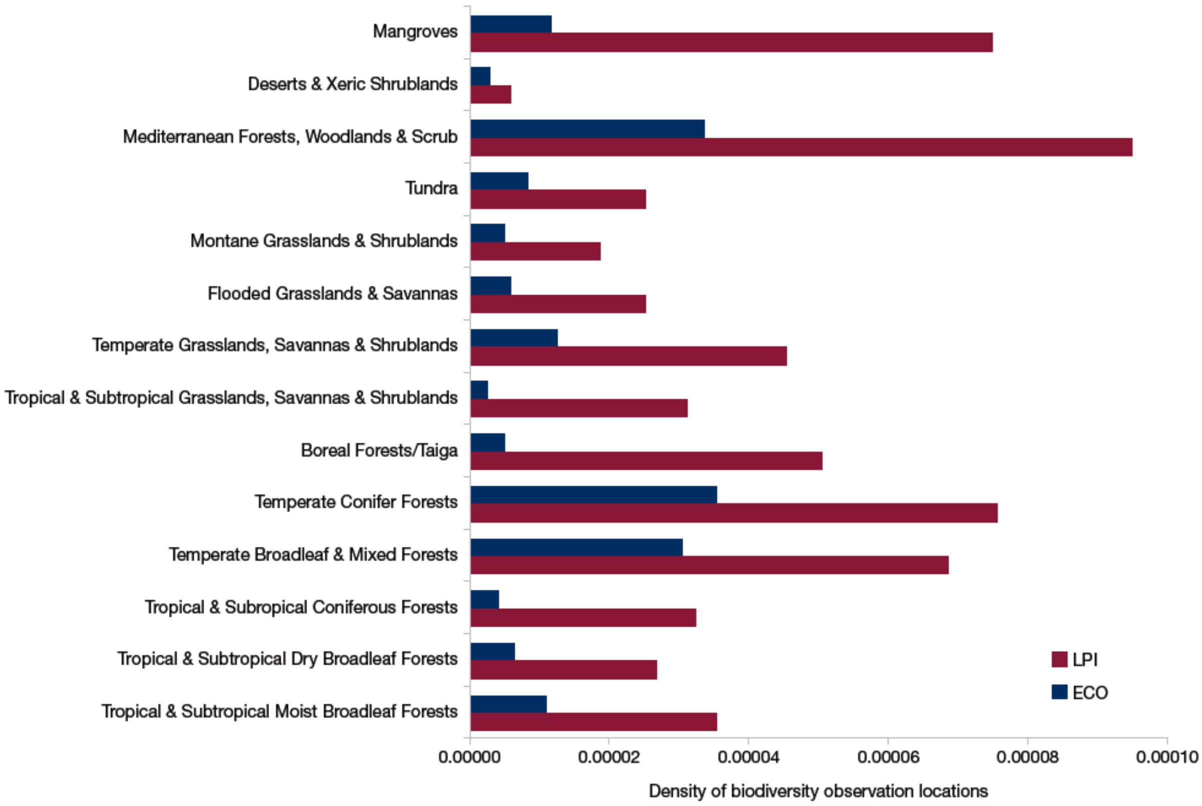



**Figure 8.2**: Number of observations per square kilometre calculated for each terrestrial biome (Olson et al., 2001). Red: Living Planet Index (LPI) study sites for 10,000 vertebrate populations with population trends collected between 1970 and 2010 (Collen et al., 2009). Blue: Ecological studies (ECO) reported in the literature for 2,573 sites between 2004 and 2009 (Martin et al., 2012). See also Figure 7.2 for another illustration of regional bias in biodiversity studies.

## 8.2.3 Facilitating data access for model calibration and validation

Good practices in sharing data, developing open source databases and platforms, and documenting data access procedures need to be encouraged within the scientific community.

### 8.2.3.1 Improving data sharing

There is currently a major movement towards 'open data', reflecting an increasing interest in and demand for data to be made publicly available (Reichman et al., 2011; Molloy, 2011). CBD Aichi biodiversity target 19 emphasises that biodiversity information needs to be 'widely shared and transferred, and applied.' In coming years, data release is expected to be more often required by funding sources and research journals, and it will become a common norm of conduct of scientific societies. Note that this is not just a response to increasing calls for transparency from stakeholders; archiving data in public domains can potentially yield multiple benefits to the scientific community and the data providers. The opening-up of data not only helps reduce the duplication of work needed for data collection but also facilitates scientific exploration (Rüegg et al., 2014; Hobern et al., 2013) and helps address conservation problems. Considering that combining past inventory data with present data can serve as a surrogate for long-term monitoring (e.g. estimating a temporal change in species distribution in response to climate change; (Moritz et al., 2008), the digital mobilisation of existing data is crucial. This applies not only to data on natural systems, but also to social data on all aspects of human activities relevant to the status of, and pressures on, biodiversity and ecosystem services. Similarly, local and indigenous communities are sometimes the only repositories of historical data, and it is important to promote the uptake and publication of traditional knowledge (see Sections 7.3.2, 7.4.2 and 7.6.5).

Creating large datasets spanning several temporal, geographical and biological scales – essential for global assessments – requires numerous inputs from a large number of contributors. However, such broad-scale sharing can present challenges. Field data, which are the crucial part for the majority of models, need enormous effort to be collected. Therefore, data are undoubtedly precious and some people may feel reluctant to submit their data to public domains. Local communities may fear sharing their traditional knowledge because of concerns about knowledge misuse and loss of intellectual property (see Section 4.2.3).

For scientists, incentives for data sharing, including career rewards, are important to ensure the further development of data archives (Borgman, 2012; Costello et al., 2013; Hobern et al., 2013). While the potential benefits of open data have been extensively discussed in the literature, not enough emphasis has been placed on crediting and rewarding aspects of providing data. Advocates for opening up data tend to stand on the side of the 'data user', and do not necessarily view the issue from the side of the 'data collector'. According to a survey, the most dominant answer from data collectors regarding a condition for the use of data is formal citation (Michener et al., 2012). Importantly, the advent of the Digital Object Identifier for data and the encouragement to list data sources in reference lists are major factors that promote the release of data. Despite this, some data collectors may instead prefer to openly publish only the metadata. However, conflicts exist as raw data are often required by the data users. Archiving data as metadata requires users to resort to multiple, sometimes lengthy, procedures to access raw data.

Given the 'top-down pressure' (Molloy, 2011) for open data, the development of additional incentives and initiatives will be necessary for shortening the time for data to become available for models and scenarios. In this regard, inviting data collectors to be involved in data analysis may potentially help, as data collectors have first-hand knowledge about the strengths and weaknesses of the data. This co-development and collaboration between data collectors and users may benefit both, leading to 'win-win solutions'. This is one possible way of overcoming the issue, but it will not provide an ultimate solution because it may not be feasible to include all data collectors as co-authors, or possible to coordinate an analysis with potentially large numbers of people. In summary, data collectors should be encouraged to publish their data on open repositories.

Lastly, those who are involved in constructing and maintaining web interfaces and large-scale repositories have not always been well acknowledged. However, they are a critical part in scientific communities for supporting data accessibility and facilitating data users. Importantly, a rapid expansion in policy and requirements for data depository may come with the heterogeneity in data quality. To prevent noisy or poor-quality data from being archived, database managers are likely to play more important roles in the future. While a stringent set of criteria and protocols will be also required to maintain data quality, those who contribute to this process need further recognition.

#### 8.2.3.2 Accessing and using data

Both biodiversity and ecosystem services data are increasingly being made publicly available (e.g.

Boxes 8.3 and 8.4). In using such data, an important issue is data standardisation. Models and scenarios often require multiple data types, sourced from different databases. Combining data from multiple sources may be difficult; for example, biodiversity information such as taxonomic names are often stored in different ways or following different published taxonomies. Work has been ongoing to create a comprehensive formal taxonomic classification and to create architectures that can handle multiple taxonomies (Hobern et al., 2013). A number of tools are available to unify data from different sources, such as the Global Names Architecture, which can help match and integrate names of species from different sources (http://globalnames.org).

The licensing form of data also needs to be considered. For instance, many institutions make data available as open access for non-commercial use; however, data licensing policies for commercial use may have some restrictions or require a fee for usage (e.g. Creative Commons multiple licensing modes). New frameworks that help retain currency and attribution back to the original data sources will also be important to strengthen the direct linkage between data collectors and users. Another issue is that the operability of data is different between databases and between data types, largely limiting the direct application of existing data for model calibration and validation. Considering the increasing visibility of data, platforms that facilitate user access will play a crucial part in the coming years (Box 8.3). While biodiversity information such as that archived in the Global Biodiversity Information Facility (GBIF; http://www.gbif.org), in the International Union for Conservation of Nature (IUCN) Red List of Threatened Species (http://www.iucnredlist.org) and in the Ocean Biogeographic Information System (http://www.iobis.org) are widely recognised and relatively well organised, data for ecosystem services tend to be collected individually and more diversely. The difficulty of coordinating the development of repositories for large databases for ecosystem services results from the lack of common and agreed language, definitions and framework on ecosystem services.

Generally, ecosystem services data are produced by combining datasets sourced from multiple databases into a focal type of data (Tallis et al., 2012; EC, 2014). These datasets are diverse and can be physical, biological and social, such as satellite images, digital elevation models, Light Detection and Ranging (LiDAR) data, land/ocean-use information, crowd-sourced data (e.g. for taxa distribution and phenology), meteorological data, human health statistics, cultural/religious information and economic/financial statistics. Another reason why these diverse datasets are required is that, in real-world decision making, it is important to identify trade-offs and synergies between multiple services (e.g. Brandt et al., 2014; Bateman et al., 2013). Although some tools to facilitate data use are now becoming open and available (Chapter 7), handling such different datasets needs multidisciplinary and interdisciplinary skills and knowledge that are not owned by the majority of users. At the local scale, the shortage of human resources can be as serious as data incompleteness. Another issue that needs to be addressed is that of cultural values, which are heterogeneously distributed across the globe. Localised information such as traditional knowledge, which would be tightly associated with cultural ecosystem services, has not been well archived.

Some synthesised information that would potentially facilitate the non-expert use of ecosystem services information is currently available online. For example, the Ecosystem Service Valuation Database of The Economics of Ecosystems and Biodiversity (TEEB) (http://www.fsd.nl/esp/80763/5/0/50) gives a global overview of the estimates of monetary values of ecosystem services, potentially benefiting local stakeholders who are unfamiliar with environmental economics. Another example is the Global Forest Change (http://earthenginepartners.appspot.com), which makes it possible for groups without remote sensing expertise to visualise and assess the changing status of forest coverage in a specific region of interest (Hansen et al., 2013). Although such frameworks for increasing the availability of ecosystem services data are currently emerging, a comprehensive ecosystem services database would require collaboration among relevant organisations, including IPBES.

In addition to open data, open tools are also becoming increasingly numerous and available. However, it is crucial to assist different users in the use of diverse datasets. In this regard, it is desirable to expand opportunities for learning how to handle different types of data, including online learning modules and webinars that can be accessible worldwide. Many organisations, universities and research institutes now provide various databases; in addition to the information regarding the types of available data, they could also be encouraged to provide documentation and tools on how to use these data (also see Chapter 7). The growing appreciation of the need to communicate science and access information in all fields is likely to make such developments easier.

**Box 8.3:** Examples of good practices in sharing biodiversity data at the species level

A. Databases of occurrences, trends and threats

- GBIF: occurrence data -- http://www.gbif.org
- IUCN Red List: threat category, range map and information on population, trends, ecology, distribution, threats and conservation measures -- http://www.iucnredlist.org/
- Global Population Dynamics Database: time series of population abundances or indices -- http://www3.imperial.ac.uk/cpb/databases/gpdd
- North American Breeding Bird Survey: population trends and relative abundances of North American bird species -- https://www.pwrc.usgs.gov/bbs/
- Map of Life: trends and occurrence data -- http://mol.org
- Global Invasive Species Database: native and invaded ranges -- http://www.issg.org/database/welcome/
- WoRMS: taxonomy and distribution of marine species -- http://www.marinespecies.org
- OBIS: Ocean biogeographic information system: occurrence data -- http://www.iobis.org
- EOL: Encyclopedia of Life -- http://eol.org
- AlgaeBASE: taxonomic and distribution data on algae species -- http://www.algaebase.org

B. Databases of demography and life history characteristics

- TRY: Plant Trait Databases -- https://www.try-db.org/
- COMPADRE: matrix (demographic) models for plant and animal species -- http://www.compadre-db.org
- MAPS: Monitoring Avian Productivity and Survivorship -- http://www.birdpop.org/pages/maps.php
- BROT: plant trait database for Mediterranean Basin species -- http://www.uv.es/jgpausas/brot.htm
- AnAge: database of traits such as longevity, body size, age of first reproduction, etc. for animal species -- http://genomics.senescence.info/species/
- PanTHERIA: life history, ecology and geography of extant and recently extinct mammals (Jones et al., 2009; http://esapubs.org/archive/ecol/E090/184/)
- FishBase: size and other biological information on fish -- http://www.fishbase.org/home.htm
- SeaLifeBase: size and other biological information on marine species --

http://www.sealifebase.org
- EltonTraits (Wilman et al., 2014): foraging ecology of birds and mammals -- http://www.esapubs.org/archive/ecol/E095/178/

**Box 8.4**: Examples of good practices in sharing ecosystem services and biodiversity data at the ecosystem level

A. Biodiversity, ecosystems and environmental databases
- BISE: Biodiversity information system for Europe; collection of databases on biodiversity and habitat types -- http://biodiversity.europa.eu
- EcoDB numerical data of gas fluxes and micrometeorology in agricultural fields, wetlands and grasslands -- http://ecomdb.niaes.affrc.go.jp
- NOAA National Centers for Environmental Information -- https://www.ncei.noaa.gov/
- Sea Around Us: information about fisheries and fisheries-related data -- http://www.seaaroundus.org
- EDGAR: Emissions Database for Global Atmospheric Research -- http://edgar.jrc.ec.europa.eu/
- ACP Environmental observatory -- http://acpobservatory.jrc.ec.europa.eu/
- EFDAC: Europe Forest resources database -- http://forest.jrc.ec.europa.eu/efdac/
- TreeBASE: a database of phylogenetic information -- http://treebase.org/treebase-web/
- Global Land Cover Characterization -- http://edc2.usgs.gov/glcc/

B. Ecosystem services and management databases
- MESP Marine Ecosystem Services Partnership: information on the human uses of marine ecosystems around the world -- http://www.marineecosystemservices.org
- Ecosystem-based management tools: information about coastal and marine planning and management tools -- http://www.ebmtools.org
- FAOSTAT: time-series and cross-sectional data relating to food and agriculture -- http://faostat3.fao.org/
- ESP: The Ecosystem Services Partnership: a database on monetary values of ecosystem services -- http://www.fsd.nl

# 8.3 Improving models

## 8.3.1 Basic research to fill thematic gaps and build functional linkages

A wide variety of approaches to scenario analysis and modelling can now be used to inform the assessment of status and trends, to assess future risks, and to evaluate policy options (Chapters 3, 4 and 5). Despite recent advances in these approaches, there are significant gaps, both in the types of models for analysing and forecasting different ecological processes (at all levels of organization, from individual to ecosystem) and in linkages between different types of models.

This section focuses on basic science needs, in other words research directed towards the further development of theoretical and conceptual underpinnings of ecological and social-ecological systems.

Most research of this type is included in the basic science research carried out by academic scientists in various disciplines. This section gives examples of research that would advance scenario analysis and modelling in contexts and at scales of interest to IPBES.

#### 8.3.1.1 Thematic gaps

There is a need for research that leads to the development of new types of models to analyse and forecast ecological processes and ecosystem services that have so far not been the focus of much research. In this section, we give a few examples of these 'thematic gaps'.

***Species interactions and community dynamics***

Models for performing scenario analyses and projecting regional biodiversity dynamics under IPBES will need to incorporate species interactions and community dynamics (including, for example, trophic interactions and disease dynamics). There is already much progress in this area in marine systems, especially at the community and ecosystem levels (Fulton, 2010). For example, the Ecopath with Ecosim (EwE) model (Christensen and Walters, 2004) combines trophic relationships, environmental indicators and biomass dynamics in the marine environment at a range of scales, from local to global. The model also incorporates the spatial and temporal dynamics primarily designed for exploring the impact and placement of protected areas. It can be used to evaluate past and future impacts of fishing and environmental disturbances as well as management and policy options. The mechanistic General Ecosystem Model (Harfoot et al., 2014) is a process-based model that facilitates consideration of the ecological implications of human activities and decisions on both marine and terrestrial ecosystems. The model uses biological and ecological data of functional groups to explore the interactions between them and with the environment, and to make predictions about the ecosystem structure and function, ranging from the local to the global scales.

Although there is also much theoretical and empirical research on species interactions and disease dynamics in terrestrial systems and at the species level, these developments have not been translated into predictive tools at large temporal and spatial scales (Thuiller et al., 2013). For instance, while it is generally acknowledged that much of the impact of climate change will be through the disruption of existing species interactions and the emergence of new ones (Van der Putten et al., 2010), most large-scale models that project impacts of climate change on biodiversity either exclude such interactions or incorporate them only implicitly or under simplifying assumptions (Albouy et al., 2014). When species interactions are explicitly included in predictive models of biodiversity, they are often limited to only two or a few species, such as one-predator-one-prey (Fordham et al., 2013) and predator-prey-pathogen (Shoemaker et al., 2014); or they are limited to specific types of well-studied interactions such as pollination (Bascompte et al., 2006). Part of the reason for this thematic gap is that, in the context of projecting the effects of particular policy or management actions on specific systems, the challenges in community ecology are even greater than in the population ecology of single species. In other words, our understanding of the dynamics of communities is less than that of populations of single species, thus making it difficult to develop models that have sufficient skills to directly inform policies and management.

Therefore, basic science investments that lead to the incorporation of species interactions and community dynamics in scenario analysis and modelling at large spatial and temporal scales would benefit global and regional IPBES assessments. Research needs include large-scale experiments (e.g. experimental translocations), long-term and large spatial scale monitoring of the effects of conservation or policy actions (e.g. monitoring following the establishment of protected areas and invasive species control measures), and studies designed to translate measurable properties (such as a comparison of ecological niche models of potentially interacting species) into parameters commonly used in theoretical models of species interactions (such as interaction coefficients or partial derivatives of population growth equations).

Recent studies have attempted to improve the mechanistic understanding of the relationship between species diversity and ecosystem functioning by using a functional group (trait) approach instead of species richness. In terrestrial environments, a comparison between a trait-based approach and a taxonomic approach indicated that ecosystem functioning was predicted better by the trait composition than by the number or abundance of species (Gagic et al., 2015). However, a review of over 110 experimental studies has shown that richness is positively associated with ecosystem function (Cardinale et al., 2006). An increase in species richness increases the ability of that functional group to exploit and deplete resources, such as primary space, food or nutrients, which has usually been considered an indication of 'ecological performance' (Wieters et al., 2012). The diversity of these results would suggest that new modelling approaches that integrate biodiversity composition and ecosystem function are required, to achieve an improved understanding of ecological systems and provide more accurate predictions of future states and management outcomes.

***Early warning of regime shifts***

Another research need is the development of practical early warning systems to anticipate ecological breakpoints, tipping points and regime shifts (Leadley et al., 2014a). Although much research has been done on regime shifts in ecosystems, there are significant gaps, with the result that no practical early warning system for regime shifts (i.e. a set of generally agreed-upon measurable indicators) is currently available for adoption by IPBES. While generally agreed-upon indicators may be desirable, they may not be possible given system specificity. Practical limitations include dependence on long-term time-series data (which are not as practical as static measures, such as spatial patterns often used at the species level), the difficulty of determining critical thresholds for a specific ecosystem, the difficulty of predicting the timing of the transition and the nature of the altered state.

At the species level, warning systems based on current status and recent trends of populations have been in use for decades (Mace et al., 2008), and have been recently tested under scenarios of climate change (Stanton et al., 2015). At community or ecosystem levels, warning systems based on statistical properties of time series – such as increasing temporal variance and autocorrelation, and slowdown of system recovery from small perturbations – have been proposed (Scheffer et al., 2009) and empirically tested (Carpenter et al., 2011). For example, Mumby et al., (2013) used ecological models and field data to show that coral reef systems are likely to have multiple attractors and that they can shift to and get stuck in an undesirable (degraded) alternative stable state. A promising research direction is linking theoretical research on network robustness and empirical research on indicators of resilience, which have been largely unconnected so far (Scheffer et al., 2012). A related, and also promising, research direction is using time-series data of ecological variables to infer causal drivers of ecological change. Regime shifts may be more predictable if the underlying ecological processes are understood. Methods such as maximum likelihood (Wolf and Mangel, 2008), convergent cross-mapping (Sugihara et al., 2012) and Bayesian model selection (Shoemaker and Akçakaya, 2015) have been used to infer causes of species decline and to separate causality from correlation.

The further development and refinement of existing approaches will help advance the use of mechanistic models for building early warning systems as well as for evaluating the effect of policy options on biodiversity and ecosystem services.

***Response to variability and extreme events***

One critical research need related to regime shifts, at both species and ecosystem levels, involves the effects of changes in environmental variability and environmental regimes, and biodiversity responses to extreme events (Zimmermann et al., 2009). In particular, global climate change is expected to result in the increased frequency and intensity of extreme weather events.

Predicting the effects of projected weather variability on the properties of biological systems (including their persistence and variability) requires multidisciplinary collaboration among climatologists and ecologists, as well as the integration of information from demographic models, physiological models and predictions of climatic variability.

***Developing models for projecting biodiversity indicators***

Many of the currently used or proposed indicators (see Section 8.2.1) are useful for assessing current status and recent trends of components of biodiversity and ecosystem services, but few can be projected into the future. Research that links indicators and modelling can fill this gap.

Such research would allow for the simulation testing of indicators to evaluate their reliability and information content, which also supports the identification of indicators that can be used to not only measure the current status, but also to forecast the future state of biodiversity and ecosystem services, based on scenario analysis and modelling. One key research direction is developing models that can project future values of biodiversity indicators for alternative policy options. For instance, in marine systems, size-based models generate simulated size distributions, abundance and productivity of multiple species, which are then used to calculate size-based indicators and characterise potential future ecosystem states under alternative management options (Blanchard et al., 2014). Another example is the IUCN Red List threat category, a biodiversity indicator of species-level extinction risk, which has been projected under scenarios of climate change using coupled niche-demographic models (Stanton et al., 2015).

***IPBES-relevant scales***

Most basic ecological research involves short time periods and small spatial scales, which would be relevant to short-term scenarios and local scales. However, they may not be relevant to the long-term scenarios for the global and regional assessments to be undertaken by IPBES. There is a need for investment in research on ecological processes at the spatial and temporal scales relevant to IPBES assessments. This is especially important for regional assessments, both because IPBES will undertake them first, and because global assessments will need data and model support from sub-global assessments to fill knowledge gaps. In addition, there is a bias in the taxonomic and regional coverage of basic research, with a disproportionate amount of research involving the populations of a few groups (such as birds and mammals) and focusing on certain regions (such as northern temperate regions). There is also a need for academic modellers and ecologists to become more familiar with applied fields such as forestry, fisheries and agriculture, where policy-relevant models have been used at scales relevant to IPBES (e.g. Platts et al., 2008; Blanchard et al., 2012; Kok et al., 2014).

#### 8.3.1.2 Functional linkage gaps between biodiversity, ecosystem services and human well-being

There is a research need to develop linkages concerning functionality between biodiversity and ecosystem function, human well-being and natural systems.

***Coupling social and ecological models***

One type of linkage that is needed is between human socio-economic systems and natural systems. Improving the coupling of the social and ecological components of models and scenarios requires well developed, specific feedbacks from the ecological to the social systems and vice versa (Carpenter et al., 2009; Figueiredo and Pereira, 2011). Research on these matters requires not only an understanding of how people make decisions to enhance their well-being, but also an understanding of the context in which they make those choices. Moreover, it is important to consider whether information about the effects or consequences of these decisions is available and, if it is, whether it is used in making decisions. These decision processes are poorly understood but remain essential.

Linkages between human and natural systems may have complex structures and may form cascades. For example, the effect of human activities on the world's climate is fairly well studied. There are also studies on the second link, the effects that climate change have on human activities, such as shifts in agriculture and urbanisation. The third link is the effect of these changes in human activities on biodiversity and ecosystem services, compounding the direct effects of climate change on natural systems. Other examples include the linkages among human population growth, land-cover change and ecosystem services (Pereira et al., 2010; Brock et al., 2009). Such cascades of causal connections are often difficult to predict (Chapman et al., 2014; Watson, 2014).

Understanding the linkages between the ecological and the social components and identifying the underlying feedbacks and cascades are vital to understanding the dynamics of the coupled system. Understanding how people perceive that their well-being is affected by environmental conditions, how policies are designed and accepted, and how people may change their behaviour as their environment changes are essential components of scenario modelling (Perrings, 2014). Moreover, an understanding of how values vary between individuals and groups, how they relate to context and scale and how they change with time is crucial for assessing nature's benefits to people and human well-being.

The modelling communities in the natural and social sciences are relatively isolated from each other, and a substantive collaboration effort is needed. Model co-design will promote intellectual fusion between communities, helping them to formalise and integrate different discourses into a consistent framework (Rindfuss et al., 2004). Such an effort will necessitate overcoming linguistic, epistemological, technical and other hurdles between the modelling communities. Moreover, in order to increase the policy relevance, including problem framing, and the transparency relating to aspects such as social justice and equality, modelling and qualitative cultural research need to be brought into the conversation.

It is therefore critical to encourage research on the coupling of human and ecological systems that focuses on these causal chains and feedbacks as well as on other relations, and on the scale at which these linkages operate, to help modellers make more adequate projections of future changes in biodiversity and ecosystem services.

Other types of coupling that are needed include those between ecosystem types, such as between terrestrial and freshwater ecosystems. A greater understanding of the functional connectivity within and between terrestrial, freshwater and marine ecosystems would help address a variety of questions related to ecosystem services, for instance in the design of diffuse pollution mitigation measures to prevent downstream eutrophication.

***Linking biodiversity and ecosystem services***

A critical research need involves the functional linkages between biodiversity and ecosystem services (Mace et al., 2012; Díaz et al., 2007). As the previous chapters have emphasised (e.g. see Chapters 4 and 6), only a limited number of models attempt to predict the impact of ecological changes on human well-being (for some examples see Pattanayak et al., 2009 and Bauch et al., 2015). Furthermore, many models and spatial assessments of ecosystem services rely on land cover and other biophysical variables such as topography, but have a limited treatment of the effect of biodiversity at the species and community levels, including much of the regional-scale work carried out in Europe (Schulp et al., 2014), or at the global scale (Karp et al., 2015). There is a need to demonstrate the role of biodiversity and ecosystem health in underpinning ecosystem services and for reinforcing the understanding of the relationships between ecological mechanisms and ecosystem services to create realistic end products for managers (Wong et al., 2015). One of the few well-developed connections is between pollinators and human well-being (see IPBES thematic assessment of pollinators, pollination and food production). A particular challenge is modelling not only the supply or potential supply of ecosystem services, but also the service actually used or enjoyed by people, which often requires assessing the demand for the service and the social preferences of communities (Tallis et al., 2012; see IPBES Deliverable 3d on the diverse conceptualisation of values). Another significant challenge is that existing models are usually one-way linked, which may not capture the non-linear dynamic linkages between different components of biodiversity and ecosystem services (e.g. see Chapter 6).

Developing such integrated models, tools and methods will require basic research involving multidisciplinary teams of scientists (including economists and social scientists, in addition to natural scientists), as well as policymakers and other stakeholders (see Section 8.4).

***Integrating process-based and correlative approaches***

Development of the types of functional linkages between different types of models of biodiversity and ecosystem services discussed above can be facilitated by research into process-based (mechanistic) as well as statistical (e.g. correlative) relationships.

For example, the analysis of statistical relationships between environmental drivers (climate, land-cover) and biodiversity components (e.g. species occurrence) allows some predictive ability. Such an approach has been successfully implemented as ecological niche models and used to project the future potential distribution of species in response to environmental change (e.g. Guisan and Thuiller, 2005). However, to predict beyond current conditions, and to evaluate the impact of management and conservation options, a deeper understanding of ecological processes is needed. This need has led to the development of more mechanistic models that incorporate ecological processes such as dispersal and demography (e.g. Keith et al., 2008) and the coupling of correlative and process-based approaches (Boulangeat et al., 2014). Similarly, the development of linkages discussed in this section is likely to benefit from coupling correlative or statistical methods with mechanistic models of ecological and socio-economic processes, such as some of the models incorporated in the Integrated Valuation of Ecosystem Services and Trade-offs (InVEST) package (Daily et al., 2009) or integrated assessment and system models.

***Platforms for model linkage***

On the technological side of developing these linkages, there is a need to encourage the development of models that can communicate with (or that can be embedded in) software platforms that are designed for linking different models.

Two main types of such platforms are 'scientific workflow managers' and 'integrated environmental modelling frameworks'. Both of these approaches allow users to assemble and run a system composed of existing simulation models that can exchange data at run time. Examples of scientific workflow managers include Kepler (kepler-project.org), with applications in areas such as ecological niche modelling (Pennington et al., 2007) and environmental sensor data analysis (Barseghian et al., 2010); VisTrails (vistrails.org), recently applied to habitat modelling (Morisette et al., 2013); and Taverna (http://www.taverna.org.uk), recently applied to mapping potential distribution patterns (Leidenberger et al., 2015). The integrated modelling frameworks include OpenMI (openmi.org), Object Modelling System (www.javaforge.com/project/oms) and Metamodel Manager (www.vortex10.org/MeMoMa.aspx), which have been applied to models of hydrology (Butts et al., 2014), sediment transport (Shrestha et al., 2013), trophic interactions (Prowse et al., 2013) and solar radiation (Formetta et al., 2013). An important difference between these systems is that the workflow managers are mainly designed for the infrequent, unidirectional transfer of data among component models, whereas the integrated modelling frameworks are designed for among-component interactions (i.e. feedbacks) and for the frequent exchange of data among modules (e.g. passing key information at every time step), thereby allowing two-way interactions between two linked models.

Other technological improvements required for integrated or coupled models include compatible spatial and temporal scales (coverage and resolution; see Chapter 6); data-based and region- or system-specific functional relationships; and interacting drivers (see Chapter 2).

#### 8.3.1.3. Evolving methodological reviews and research prioritisation

Research on many aspects of scenario analysis and biodiversity and ecosystem services modelling is progressing at a rapid rate. Many of the approaches reviewed in this report will be further developed in the near future; others may become obsolete. Therefore, there is a need to ensure – through ongoing updates and new evaluations – that the review of available policy-support tools and methodologies for scenario analysis and biodiversity and ecosystem services modelling continues to reflect best available science. Similarly, there is a need for the ongoing prioritisation of research needs. Some of the research and development directions and needs identified in this chapter will have already matured in the next few years, while others will not be pursued, or will be proven to be not beneficial.

Therefore, it is critical that IPBES develops mechanisms for research prioritisation, to encourage basic research that advances scenario analysis and modelling in contexts and at scales that are relevant to IPBES.

This could be through the IPBES Expert Group on Scenarios and Modelling (Deliverable 3c), Conceptualisation of Values (Deliverable 3d) and Policy Support (Deliverable 4c) and the Task Force on Knowledge and Data (Deliverable 1d), which could make recommendations to research funding agencies about the significant gaps that remain in our understanding of the fundamental processes that are the subject of scenario analysis and modelling used in IPBES assessments. Such recommendations would benefit from input from policymakers, resource managers and planners, both applied and academic natural resource modellers and researchers, and ecological, economic and social scientists.

### 8.3.2 Verifying and validating models

To be of any use for IPBES and other applications such as conservation planning or decision making, models and ultimately scenarios need to have a full treatment and report of uncertainty, together with a proper and sound validation.

In biodiversity and ecosystem modelling, the heterogeneity of data and the range of factors influencing the results mean that the tasks of analysis and validation can be complex. Model validation covers different approaches and goals, but the overall idea is to use a set of criteria to classify and identify an acceptable model. Agreement between model output and observed/experimental data of any sort can be analysed qualitatively using appropriate graphical design to visualise model performance. In addition, and complementary to visual validation, statistical analyses and accuracy tests are pivotal to make model validation and model comparisons robust, general and quantitative. Model validation (or assessment of model skill) is a growing topic area with existing precedents in biophysical, climate and weather modelling (e.g. for the Intergovernmental Panel on Climate Change (IPCC) see Flato et al., 2013). However, there is a lack of standardised terminology and approaches to validate biodiversity and ecosystem service models and their application for scenario building. IPBES could be the driving force to prepare such guidelines, as they are critical for users to trust models and scenarios and for developing global or regional syntheses. In this development, model pedigrees could be highly valuable tools to build trust in the output of existing and used biodiversity and ecosystem service models. Model pedigree is the measure of confidence the research community has in a given model and is influenced by factors such as the testing and verification of internal model processes; the quality of the data used; acceptance and use of the model by a large part of the community; applications of the model to a wide variety of cases, questions and taxa; the transparency and documentation of the model structure, assumptions and functions; and the scientific and technical credibility of the model developers.

A model may be general (can be useful in many different situations), realistic (parameters and variables are based on true cause-effect relationships) and precise (accurate quantitative output), but it is impossible to have a perfect model that can maximise all three of these attributes simultaneously (Levins, 1966). Models are often built to gain a deeper understanding of the interactions between system components and to respond to questions about the functioning of the systems (thus increasing 'reality'). Hence, the limitations of a model need to be assessed from the start and adequately communicated to the stakeholders who will be using the outputs. There is a need for appropriate guidelines for validation that could be applicable to a large range of biodiversity, ecosystem process and ecosystem service models. The difficulty in creating such protocols is that the variety of existing models is large and will require different strategies. The Expert Group on Scenarios and Modelling could be the leading force for such standardisation.

There are several issues modellers and users should consider when validating a biodiversity or ecosystem service model and associated scenarios.

<u>The goal of the validation</u>: There are several ways of validating a model and the appropriate approach depends on the overall purpose of the validation. The purpose of validation should therefore always be clearly defined and reported since the subsequent tests, whether they are qualitative or quantitative, will be linked to that specific validation purpose. The output of the

validation procedure gives important feedback to the modeller on how the models could be improved, but also to the end users on whether the model can be used, or with what confidence it can be used for a specific purpose. In biodiversity modelling, one may want a model that correctly predicts the equilibrium range of a species, in which case a visual inspection of observed and predicted maps and associated statistics would be sufficient. However, such a validation procedure will not give any information to the end user or stakeholder on the ability of the model to simulate the transient dynamics of species in response to a given environmental change. For such purposes, modellers require dynamic models and time series of data for validation.

Model and scenario comparison: Model and scenario comparisons should also be part of the validation procedure. For any given phenomena, several alternative models and scenarios can be developed, for instance at different levels of complexity. Comparing several models or scenarios built or calibrated for the same system and purpose allows us to: (i) understand their respective behaviour, (ii) choose the best one if needed, (iii) understand the effects of structural uncertainty on model outputs, (iv) average the models, or (v) build an ensemble forecast to visualise and apprehend the overall variation of the models and scenarios given the data and system (Araújo and New, 2007). Species range modelling is one of the areas in which statistical models and process-based models of increasing complexity can be benchmarked against observed data. Cheaib et al., (2012) compared eight different species distribution models, from purely statistical models to highly complex individual-based models, under current and future conditions. While varying the effects of environmental drivers, they singled out the assumptions made, the drawbacks therein, and the ability of these models to project the potential distribution of species (Cheaib et al., 2012). Although such evaluations and comparisons have been done in a number of studies for modelling the distribution of species (Kearney et al., 2010; Morin and Thuiller, 2009), of dynamic vegetation processes (Cramer et al., 2001), or of resulting ecosystem services (Bagstad et al., 2013), we argue that the systematic comparison of different models and scenarios and the building of model ensembles to project both trends and uncertainties should be a golden standard, as is currently the case in climate change research. Such comparisons, together with an analysis of uncertainties, are critically important if the outputs of such models are to be used for decision making or conservation planning. Ensemble modelling or ensemble forecasting is the appropriate method in this regard if paired with appropriate validations and a formulation of uncertainty.

Model predictions and scenarios: Most biodiversity and ecosystem services models are built to provide predictions based on scenarios, for instance under changing climate and land use. As such, these predictions can be compared with expert knowledge, experimental data, observed data and virtual data. A plethora of approaches and statistical techniques exist (e.g. residual mean square errors) and have already been thoroughly compared and discussed. Clear predictions, using robust statistical methods, and the generation of enough data (either experimental or observational), are pivotal elements for reaching the level of quality needed for validation. Biodiversity and ecosystem services models are often subject to data limitations because of the difference between the scale of prediction and the scale of measurement. For instance, most dynamic vegetation models use growth curves that are calibrated over dozens of individuals (e.g. trees) measured in situ with precise climate measurements. These curves are then extrapolated over large spatial scales and with resolutions such as 20x20 km for which climate is highly smoothed. The outcome can then no longer be directly compared with the growth of single individual trees. To overcome this limitation, cross-scale validation has been proposed (using data generated at a finer scale to validate models built for a larger scale). But even here, the question of interchangeability of processes between scales has not been truly addressed (Morozov and Poggiale, 2012).

Predictions involving future conditions pose special problems for validation, since the temporal scales are such that we often cannot test the validity of models in the future, which could be populated with previously unobserved phenomena. In this regard, biodiversity and ecosystem service models can be considered validated if they successfully predict past events (retrospective testing; e.g. Brook et al., 2000). However, the probability of making meaningful projections decreases with the length of the time period into the future.

A continuous exchange of validation data among developers and test teams should either ensure a progressive validation of the models with time, or highlight the need for updated interpretations of the analysed system (population, ecosystem, community or landscape). To this end, spatially and temporally dynamic models of biodiversity or ecosystem services must be validated against monitoring data.

### 8.3.3 Managing uncertainty in models

Linguistic and scientific uncertainty in models can be reduced by developing new technical approaches and by engaging stakeholders and local populations in the model development process.

With the rise of statistical and mechanistic predictive models of biodiversity and ecosystem services, quantifying, incorporating and propagating uncertainty have become key issues. Regan et al., (2002) recognised two main types of uncertainty in environmental science: scientific (also called epistemic) and linguistic (Table 8.2). As seen in Chapter 4, scientific uncertainty relates to the knowledge of the system and includes data bias and limitations, structural uncertainty, parameter uncertainty, extrapolation and interpolation, while linguistic uncertainty comes from the vague, ambiguous, imprecise and context-dependent vocabulary. The definition of a species as a unit and its general use is one simple example, and the word biodiversity is another. Although integrating linguistic uncertainty is not new in conservation biology where policy and decision making are part of the process, it is generally ignored in most cases, and only scientific uncertainty is considered.

**Table 8.2:** Sources of uncertainty and potential treatment (Modified from Elith et al., 2002 and Regan et al., 2002. *A taxonomy and treatment of uncertainty for ecology and conservation biology.* Copyright © 2002 by John Wiley Sons, Inc. Reprinted by permission of John Wiley & Sons, Inc).

| | Source of uncertainty | General treatments |
|---|---|---|
| Scientific uncertainty | Measurement error | Statistical techniques; use of intervals |
| | Systematic error | Recognize and remove bias |
| | Natural variation | Probability distributions, intervals |
| | Inherent randomness | Probability distributions |
| | Model uncertainty | Validation, revision of theory based on observation, discussion with end-user, prediction |
| | Subjective judgment | Degree of belief, imprecise probabilities |
| Linguistic uncertainty | Numerical vagueness | Sharp delineation, fuzzy sets, rough sets, superevaluations |
| | Non-numerical vagueness | Use multidimensional measures than treat them as numerical |
| | Context dependence | Specify context |
| | Ambiguity | Clarify meaning |
| | Indeterminacy in theoretical terms | Make decision about future usage of term when need arises |
| | Underspecificity | Provide narrowest bounds |

A model is as good as the assumptions behind its construction, in other words, what is accepted as true or as certain to occur. Structural uncertainty is a key consideration when sub-models or assumptions are likely to be wrong or uncertain (see Chapter 4) and can be addressed using validation (Section 8.3.2) and by using multiple models with alternative structures.

Data are essential for developing conceptual models that will later translate into quantitative or qualitative models, and also for calibrating and evaluating those models. When the information is incomplete, unreliable, imprecise, fragmented, contradictory or in any way deficient, it is fundamental that stakeholders understand that even a simple model based on very general data can be useful for providing insight into the possible effects of different alternatives. In addition, there are diverse mathematical or statistical techniques that can deal with information deficiencies, including fuzzy inference systems and uncertainty-based information theory (Klir and Bo, 1995; Cao, 2010). One advantage of fuzzy inference systems is that they allow for the incorporation of qualitative information that local experts and stakeholders may volunteer to provide. This information may then be integrated into a more rigorous framework of model construction. Qualitative reasoning helps in the construction of *knowledge models* that capture insights from domain experts about the structure and functioning of the system (Recknagel, 2006). Artificial neural network models may also be helpful in situations in which a response variable should be estimated or its behaviour predicted as a function of one or several predictor variables. Artificial neural network models have been conceptualised as non-parametric statistical techniques because they do not require the fulfilment of the theoretical assumptions of parametric statistics. They are also considered as non-linear regression techniques.

The input data for biodiversity and ecosystem services models and scenarios are often uncertain and specified as a range of values or as statistical distributions. Uncertainty analysis aims to quantify the overall uncertainty of model results in order to estimate the range of values that the output could take (Regan et al., 2002). In recent years, there has been an increasing interest in uncertainty analyses, partly motivated by the goal of keeping imperfect data in data-poor model environments instead of discarding them. Uncertainty and dependence modelling, model inferences, sampling design, screening and sensitivity analysis and probabilistic inversion are among the most active research areas (Kurowicka and Cooke, 2006). To date, despite few positive examples and the awareness that different algorithms are likely to result in different projections, biodiversity and ecosystem services models are too often used without the clear reporting of the underlying uncertainty in parameter estimation or the uncertainty resulting from the input data (see Section 4.6.1).

The better integration of statistical analyses into the parameter estimation of mechanistic models could foster the appropriate characterisation and reporting of uncertainty. Promising approaches for doing so include inverse modelling or Bayesian computation, which produce a probability distribution of the estimated parameters (the posterior distribution) that are relevant for the reporting of uncertainty (Hartig et al., 2012). So far, however, a full treatment of uncertainty has been considered too time-consuming and complex to be achieved in biodiversity and ecosystem services models, and the full integration and partitioning of the uncertainty originating from different sources (such as climate or land-use models) is difficult to achieve. To meet this challenge, there is a need for mathematical, statistical and computational skills that extend beyond the range of standard ecological expertise, and that include novel techniques mixing deterministic and random concepts that are usually considered as independent skills and expertise. For instance, Bayesian calibration, comparison and averaging can be used in biodiversity and ecosystem service models to be used in IPBES assessments. These methods require the capacity to integrate process and parameter uncertainty and incorporate prior, even qualitative, knowledge. These approaches have mostly been tested with forest-gap models (Van Oijen et al., 2011, 2013), but they could certainly be extended to many other types of biodiversity and ecosystem service models.

Pragmatic approaches are encouraged, for instance by sub-sampling alternative climate projections for the same scenario to obtain a basic representation of the uncertainty; or by considering that parameters in mechanistic models should not be fixed to one value but rather sampled from probability distributions representing uncertainty. While climate research has been producing such ensemble projections for some time (e.g. the World Climate Research Programme's (WCRP) Inter-Sectoral Impact Model Intercomparison Project (ISI-MIP)), this is not often done in biodiversity models (e.g. land-use models). This situation poses serious challenges when modellers have an ensemble of climatic data and only a few discrete scenarios of land use as input for deriving biodiversity scenarios into the future.

## 8.4 Improving scenarios and policy support

Scenarios play a major role in assessments by helping decision makers explore the impact of a broad range of policy options and socio-economic pathways on biodiversity, ecosystem services and human well-being. Quantitative models are one of the main tools used in scenarios to assess such impacts. In this section, we identify areas for improving scenarios in biodiversity and ecosystem services assessments at each step of the scenario development iterative cycle (Figure 8.1). We first examine how best to engage stakeholders in scenario development. Next, we discuss how to improve the links between models and policy options in scenarios. We then examine how the results of scenarios can be better communicated to policymakers and other stakeholders and, finally, we propose avenues for improving the impact of scenarios in decision making.

### 8.4.1 Engaging stakeholders and identifying policy needs

Identifying and engaging stakeholders in the scenario development process is essential to identify policy options. Encouraging stakeholders to participate in models and scenarios from an early stage fosters mutual understanding and trust and empowers participants with respect to the assessment goal. A key policy issue is to manage trade-offs and also opportunities for synergies between biodiversity conservation, food security and livelihoods across contrasting social-ecological regions.

'Stakeholders' are any individuals, groups or organisations that affect, or could be affected by (whether positively or negatively), a particular issue and its associated policies, decisions and actions (Grimble and Wellard, 1997; Lucas et al., 2010). 'Actors' are active stakeholders who influence the process, while 'users' are stakeholders who use the products of an assessment, such as decision makers. The early engagement of stakeholders in scenario development is crucial to enhance the legitimacy, salience and credibility of an assessment (Cash et al., 2003; UNEP et al., 2009). *Legitimacy* means that the relevant stakeholders are included in the assessment and perceive the process as unbiased and meeting standards of political and procedural fairness (Cash et al., 2003; UNEP et al., 2009; Lucas et al., 2010; TEEB - The Economics of Ecosystems and Biodiversity, 2013). *Salience* means that the assessment must be relevant by addressing problems relevant to the users (instead of, for instance, questions mainly relevant to the researchers), and that it takes into account the ecological, governance or legal context of the issues. *Credibility* means that the stakeholders are willing to accept the results of the assessment.

As the number and/or variety of stakeholders increases, conflicts of interest are more likely to occur, especially with regard to the engagement of private sectors (Hochkirch et al., 2014). The inappropriate selection of stakeholders causes loss of legitimacy by excluding agents of interest groups, and decreases relevance and credibility. 'User needs assessment' and 'stakeholder analysis' are recommended methods to adopt at the beginning of the assessment for this purpose (Hesselink et al., 2007; Grimble and Wellard, 1997). Stakeholder analysis is especially useful to ensure that under-represented categories are included, such as the 'chronic absentees' or 'hard-to-reach' stakeholders (Padovani and Guentner, 2007). Stakeholder analysis can be structured according to five steps: (1) define the context affected by a decision or action (see Section 2.2), (2) identify all stakeholders at the different scales of the assessment, (3) identify their interests, (4) differentiate and categorise the stakeholders, and (5) investigate the relationship between stakeholders. In identifying and recruiting stakeholders, transparency of the process should be maintained such that all stakeholders have the opportunity to be heard and to participate (TEEB - The Economics of Ecosystems and Biodiversity, 2013).

A range of participatory methods and tools have been proposed to engage stakeholders in co-designing scenarios (Box 8.5). Participatory scenario development can be used to improve the transparency and relevance of policymaking, by incorporating the demands and information provided by each stakeholder, and to negotiate outcomes between stakeholders. Models allow for the comparison of multiple options and the easy substitution of alternative assumptions, while also making trade-offs and potential conflicts of interests between stakeholders explicit (TEEB - The Economics of Ecosystems and Biodiversity, 2013). Cultural diversity among stakeholders, including indigenous and local communities, may bring up multiple possible interpretations of a situation (Sections 2.2.1 and 7.4.3; Brugnach and Ingram, 2012). Stakeholder interactions become essential to create a shared understanding of a situation. In this way, decision choices become the direct product of shared rules, agreements and practices developed from working together (Section 5.3; Brugnach and Ingram, 2012). Hence, research efforts need to be oriented towards integrating and producing knowledge in an inclusive manner.

A key policy issue is how to manage trade-offs and opportunities for synergies between biodiversity conservation, food security and livelihoods across contrasting social-ecological regions. In particular, the research community needs to: i) identify the nature of these trade-offs and synergies across social-ecological systems and regions of the world; ii) identify the key ecosystem services that are at stake in these trade-offs; iii) identify the biophysical and societal drivers that contribute to exacerbating the trade-offs and those that contribute to reducing them; and iv) identify opportunities for synergies between biodiversity conservation, food security and livelihoods that are most suitable for particular social-ecological contexts (Klapwijk et al., 2014; Smith et al., 2013; McCarthy et al., 2012).

**Box. 8.5:** Participatory scenario development

Participatory scenario development allows for the integration of stakeholders' values and visions in the scenario formulation as well as in the framing of scenario assumptions (Börjeson et al., 2006; Shaw et al., 2009; Forrester et al., 2015). There are different approaches for implementing participatory scenarios, ranging from time-demanding truly bottom-up processes of storyline development (Carvalho-Ribeiro et al., 2010; Sheppard, 2005) to more expedited approaches such as 'confronting' stakeholders with a storyline already developed as a prompt for discussion (Van Berkel et al., 2011). Independent of the method used, stakeholders must have the opportunity to represent their own interests and knowledge in the scenario storylines in such a way that they feel rewarded by their engagement in the scenario development process (Flyvbjerg, 2001). Because, in general, stakeholders can judge trade-offs and assess the ways in which land change affects their livelihoods, participatory scenarios can play an important role in addressing the linkage gaps between biodiversity, ecosystem services and human well-being (Section 8.3.1.2). Local and regional stakeholders can also provide insights into the role of spatial variation in the delivery of multiple ecosystem services (Van Berkel et al., 2011). Participatory scenarios are therefore particularly well suited for gaining a richer understanding of trade-offs among possible biodiversity futures (Carpenter et al., 2006). Despite wide agreement on the advantages of participatory processes, there are also shortcomings related to the effects of 'powerful' stakeholders who may strongly influence participatory processes. Implementing participatory scenarios also requires time for resolving conflicts, to account for possible shifts in policy and economic conditions as the participatory process evolves. One of the tools that has proven useful for comprehensive stakeholder engagement is visualisation techniques (Vervoort et al., 2010; Appleton and Lovett, 2003), which can improve communication efficacy by ensuring that everyone is operating in the same context (see Section 8.4.3.1).

Local communities and indigenous peoples have a wealth of traditional knowledge and are valuable sources of information (see Sections 4.2.3 and 7.3.2; Pert et al., 2015). In these communities, the knowledge of the ecosystems and their resource use and conservation practices are related to cultural aspects and religious beliefs (Section 7.4.3; Gadgil et al. (1993). This means that people in these communities may not trust persons outside their community sufficiently to share their knowledge. Overcoming this requires the development of participation channels through the work of anthropologists and social scientists, and efforts should be made to systematically gather and organise such information. There are some lessons to be learned from climate science and efforts to include traditional ecological knowledge in mitigation and adaptation strategies (Dewulf et al., 2005; Smith and Sharp, 2012; Brugnach et al., 2014). IPBES Deliverable 1c is set to provide guidance on procedures, approaches and participatory processes for working with ILK systems, while IPBES Deliverable 1c considers different approaches as well as procedures for working with ILK in assessments of biodiversity and ecosystem services. It is clear that research is needed on developing robust methods to elicit ILK that is, in many situations, key to the development of models and scenarios (Hesselink et al., 2007).

## 8.4.2 Linking models to policy options in scenarios

Short-term scenarios can be used to assess policies that act on direct drivers. Long-term scenarios are needed to assess policies that act on indirect drivers or to assess long trajectories of direct drivers. Regional IPBES assessments can use short-term scenarios or existing long-term socio-economic scenarios, while the global IPBES assessment could foster a new generation of long-term scenarios.

Scenarios can be developed using a variety of approaches (Kok et al., 2011; Alcamo, 2001) and can be categorised in two broad classes: exploratory scenarios and policy intervention scenarios (Sections 1.3.2 and 3.2.2). In exploratory scenarios, the analysis starts in the present and different plausible future trajectories are explored by stakeholders, often across major axes of uncertainty on social-ecological dynamics, and using associated narratives for the unfolding of events from the present to the future (Kok et al., 2011; Alcamo, 2001). Exploratory scenarios are often associated with the problem identification stage of the policy cycle (Section 3.2.2), and examples include the MA and the IPCC Special Report on Emissions Scenarios. In policy intervention scenarios, the goal is to assess how specific policy interventions will change the social-ecological trajectories or futures (Van Vuuren et al., 2012b). These can be further divided into target-seeking scenarios and policy-screening scenarios. In target-seeking scenarios, stakeholders agree on a desirable future and then perform a backcasting analysis to identify policy interventions that may lead to the target future (Kok et al., 2011). For example, the Roads from Rio+20 scenarios (Van Vuuren et al., 2012a) defined a vision for biodiversity in 2050, then examined three pathways, each with its own set of policy options, that can lead to that vision. In policy-screening scenarios, a policy, or set of policies, is applied and an assessment of how the policy modifies the future is carried out. For instance, the Rethinking Global Biodiversity Strategies scenarios (Ten Brink et al., 2010) consider a set of policy options aimed at reducing biodiversity loss, such as an increase in protected areas, changes in diet and improved forest management. The effects of implementing those options on biodiversity are then assessed over time.

Exploratory scenarios foster creative thinking and the exchange of viewpoints between different stakeholders, but do not always provide clear actions for implementation by decision makers to reach desirable outcomes. Policy intervention scenarios are more likely to provide clear policy pathways but have been criticised for being value-laden. Some scenario exercises have tried to combine elements of both approaches (Kok et al., 2011). The scenarios used in the 5$^{th}$ Assessment Report of the IPCC defined plausible relative concentration pathways of greenhouse gases to achieve different target levels of radiative forcing for the end of the century (Moss et al., 2010; Van Vuuren and Carter, 2014). Then, emission pathways and a range of exploratory socio-economic pathways (SSP) were developed (Van Vuuren and Carter, 2014).

Scenarios can also be classified according to their temporal horizon into short-term (e.g. up to a decade) and long-term (decades to a century), addressing different policy development needs (Leadley et al., 2014b). Long-term scenarios are useful for assessing policies that act on indirect drivers, such as population growth, with dynamics that play out over large time scales and which impact direct drivers, such as land-use change. For instance, a change in fertility rates today will have the most noticeable demographic impacts in a generation. Those changes will then impact the long-term future trajectory of land-use requirements to feed the population, which in turn will impact biodiversity and nature's benefits over those long time scales (Pereira et al., 2010). In some instances, it is the biophysical system that has slow dynamics or time lags. For instance, the dynamics of the climate system are so slow that only long-term analysis can provide meaningful projections of the climate impacts of current policy changes in fossil fuel use (see Table 8.3).

**Table 8.3:** Policy applications and development pathways for long-term and short-term scenarios.

| Type of scenario | Policy application | Options available for development |
|---|---|---|
| Long-term scenarios | a. Assessing policies that act on indirect drivers<br>b. Exploring possible futures | i. Use existing indirect driver and/or direct drivers scenarios, and project impacts on biodiversity and ecosystem services. Feasible for regional assessments<br>ii. Develop scenarios for indirect drivers associated with uncertainties or specific policies and carry out full modelling cycle. Feasible for the global assessment |
| Short-term scenarios | Assessing short-term policies on direct drivers | Model direct driver impacts on biodiversity and ecosystem services under different policies. Users may only want to know endpoints, not the trajectories |

We can envision two different approaches to developing long-term scenarios in IPBES assessments (Table 8.3). One approach is to develop novel socio-economic scenarios and carry out the complete modelling cycle from indirect to direct drivers, to biodiversity and finally to ecosystem services (Pereira et al., 2010). The socio-economic scenarios could be developed around uncertainties on drivers that are relevant to biodiversity and ecosystem services (corresponding to exploratory scenarios), or with specific policies on indirect drivers with impacts on biodiversity and ecosystem services, including those related to SDGs (corresponding to policy intervention scenarios). This approach would be feasible for global assessment, but the scenario development would probably start before the beginning of the global assessment as the full scenario development cycle can take up to five years, a bit longer than the length planned for a global assessment. This approach would also allow for the closing of the feedback loop from ecosystem services to human well-being to indirect drivers in the scenario development (Pereira et al., 2010).

A simpler and faster approach that could be used by regional assessments is to resort to existing long-term scenarios for indirect drivers or socio-economic pathways (e.g. MA, IPCC SSP). Policies to be assessed could be matched to the different pathways (e.g. a policy promoting low fertility could be matched with an MA or IPCC scenario where fertility is low). In some cases, existing projections of direct drivers (e.g. land-use change or climate change) associated with those pathways can be used to assess impacts on biodiversity and ecosystem services using models or expert knowledge and downscaling techniques (Sleeter et al., 2012; Walz et al., 2014). Downscaling existing global projections to the regional scale can improve the spatial resolution of the projections and their relevance for the analysis of biodiversity impacts and decision support (Section 6.4.1).

Short-term scenarios can also be useful for assessing how policies on direct drivers affect biodiversity and ecosystem services in the short term (Leadley et al., 2014b). Short-term scenarios do not require modelling the temporal dynamics of indirect drivers or of their impacts on direct drivers. Instead, they use simple projections of direct drivers under different policies or actions (corresponding to target-seeking or policy-screening scenarios) and assess alternative futures for biodiversity and ecosystem services. Trajectories can be irrelevant as users may only want to know the endpoints of direct drivers and to assess their impacts on biodiversity and nature's benefits. Short-term scenarios can use optimisation tools to find the best actions to achieve a given target, models to assess the biodiversity and ecosystem services consequences of different land-use configurations, or simple statistical extrapolations under different policies. For instance, in systematic conservation planning, optimisation tools are used to find the minimum number of protected sites needed to achieve a given target scenario for biodiversity conservation (Sarkar et al., 2006). Ecosystem service models can be used to assess the impacts of short-term land-use scenarios on ecosystem services (Nemec and Raudsepp-Hearne, 2013). Short-term land-use scenarios can be developed through participatory exercises, using maps, photographs and Geographic Information System (GIS) tools (Carvalho Ribeiro et al., 2013; Van Berkel et al., 2011). Finally, simple extrapolations for future values of biodiversity or ecosystem services indicators under a specific action relative to current trends can be made (Leadley et al., 2014b). This range of short-term scenario techniques can be useful for global, regional and sub-regional assessments.

### 8.4.3 Improving the communication of results

The effective communication of model limitations, assumptions and uncertainties, as well as the implications of model outputs, especially probabilistic ones, is essential for the constructive use of models in decision making.

#### 8.4.3.1 Understanding model outputs and limitations in their scope

Model results need to be understood within the context of the data and the assumptions. Keohane et al. (2014) identified five plausible principles to guide communication: honesty, precision of scientific findings, audience relevance, process transparency, and specification of uncertainty about conclusions. It is particularly important that the process of constructing a dialogue between scientists/modellers and stakeholders/decision makers explicitly involves communicating the weaknesses that inevitably appear regarding present knowledge and the way in which it can be used. Being clear about what the shortcomings are should permit an increase in confidence between interlocutors.

Making it clear to users what the uncertainties in the output are, what the implications are, and also all that is not implied (Janssen et al., 2005), may have a deep effect on the decision-making process. When users participate in the scenario and model development, they are able to better comprehend the relative value of the output and its meaning because of their previous understanding and involvement in the process. However, if the intended audience was not engaged in the model construction process, much more attention needs to be given to communicating the outputs in a way that minimises misinterpretation and that does not generate confusion or mistrust. In all cases, the results need to be presented in a clear, consistent and precise way, giving preference to graphic forms or to tables that summarise the main points.

New technologies in computer science and design have made it easier to represent information on processes and/or data in a graphical form, creating a visual image – usually a chart or diagram but also video clips, movement effects and interactive visualisations. These can be efficient means of communicating complex concepts in a clear and simple way, particularly among actors with different cultural backgrounds. Although scientists usually use sketches and graphs to explain ideas and results in their work environment, they do not normally have any training on how to use these visualisation techniques to better report findings to a wider, less specialised audience (McInerny, 2013; McInerny et al., 2014). Infographics and visual representations could be valuable tools to be used from the very beginning of the iterative process of scenarios and model construction and assessment involving scientists and stakeholders, facilitating the understanding of complex processes and identifying uncertainties, and thus building confidence and empowering participants. Moreover, the planning of final visual outputs can be embedded into the development and production stage of modelling and scenario activities.

The process of constructing models, proposing scenarios and analysing them as a means of learning in advance about the effects and implications of policies on biodiversity and ecosystem services is not only a technical matter. The whole process is embedded in the cultural setting of the societies that are part of those ecosystems and that use their resources. Communicating effectively with these stakeholders requires the participation of interdisciplinary professionals with diverse skills and broad intellectual capabilities, in particular social scientists who understand the institutions and the social structure in the region, helping modellers to notice relevant issues, but who can also contribute to helping society better understand and solve environmental problems. The Task Forces on Capacity Building and on Indigenous and Local Knowledge could consider the proper ways to train and involve interdisciplinary professionals in these communication processes.

#### 8.4.3.2 The importance of communicating uncertainty

A critical challenge in communicating the results of scientific research arises when those results contain uncertainties. It is highly important that the various types of uncertainties that will necessarily appear in the modelling process, as well as in the scenario analysis, be clearly communicated to all stakeholders and decision makers so that there is full understanding of the relative weight of the output, the implications and the risks involved. Uncertainties need to be set in the context of the key messages that are being conveyed, and the implications of the uncertainties need to be explained. It may also be important to offer information on how the uncertainties can be treated or dealt with. However, decisions can be made even when gaps in information appear, data are not totally reliable, or ample variability is observed and risks are identified (see Section 8.3.2).

Recent experience, mostly related to the communication of uncertainties related to climate change (Box 8.6) or to potential pandemics, has opened the way to a more systematic analysis of how people perceive the uncertainty inherent in scientific research. These problems have captured the attention of both climate and social scientists (Janssen et al., 2005; Handmer and Proudley, 2007; Kloprogge et al., 2007; Pidgeon and Fischhoff, 2011). Research communities have emerged in which people from different fields, such as climate and environmental scientists, historians, social scientists and philosophers, examine issues of uncertainty with respect to global environmental problems with the purpose of improving the capacity to discuss and weigh related policy recommendations (e.g. www.princeton.edu/piirs/research-communities/communicating-uncertainty/).

**Box 8.6:** An example of the importance of communicating uncertainty in a science-policy interface

Keohane et al. (2014) focused on the ethics of communication between scientists and policymakers on issues such as climate change. As a case study, they analysed the treatment of possible sea-level rise as a result of the melting of ice sheets in Antarctica and Greenland in the 4$^{th}$ Assessment of the IPCC. Sea-level rise can be projected using computer simulations of global climate models and by focusing on three processes: thermal expansion of the oceans, mountain glacier melt, and ice sheet disintegration via melting and dynamical loss (or the sliding of ice sheets into the ocean). Sliding is considered the major contributing factor in Antarctica; however, scientists did not have models to estimate future changes in sliding which resulted in a high degree of uncertainty in the projections. The IPCC Working Group I assessing the physical scientific aspects of the climate system and climate change (IPCC, 2007) gave an uneven treatment to this third factor relative to the other two, creating confusion with projections lacking clarity and transparency. This led to significant differences in the estimation of sea-level rise to be used in infrastructure planning by coastal communities, making it difficult to take practical, long-term steps under a risk-based approach. It can also be noted that Working Group I and Working Group II (assessing impacts, vulnerability and adaptation) chose different approaches to dealing with uncertainty.

The IPCC has provided guidance on the consistent treatment of uncertainties in a unified language (Mastrandrea et al., 2010; https://www.ipcc.ch/pdf/supporting-material/uncertainty-guidance-note.pdf), consisting of two metrics for communicating the degree of certainty in key findings. Firstly, theory, data, models and expert judgment can be presented qualitatively in terms of confidence in their validity ('limited', 'medium', or 'robust') and in terms of the degree of agreement ('low', 'medium', or 'high'). Secondly, uncertainty in a finding can be expressed quantitatively, in terms of probabilities. Following the 'Guide on production and integration of assessments from and across all scales' (IPBES Deliverable 2a), IPBES assessments are encouraged to express their findings using a four-box model of confidence based on evidence and agreement that gives four main confidence terms: 'well established' (much evidence and high agreement), 'unresolved' (much evidence but low agreement), 'established but incomplete' (limited evidence but good agreement) and 'speculative' (limited or no evidence and little agreement).

### 8.4.3.3 The need to improve the communication of probabilistic results

All biological dynamical systems evolve under stochastic forces. In a stochastic or random process there is some indeterminacy, which is a third type of uncertainty differing from scientific and linguistic uncertainty. Even if the initial condition or starting point is known, there are several directions in which the process may evolve. Translating the meaning of output from stochastic models to persons without professional or specialised knowledge in the subject often generates confusion because there is a whole set of possible outcomes and the results are given in terms either of averages or probabilities. As mentioned earlier, and depending on the context, it is advisable to use multiple models of differing complexities and types to compare the outputs and help comprehend their meaning.

Information involving probabilities is often susceptible to bias and misinterpretation, as people have different perceptions of what is really meant. For instance, different levels of comprehension of weather forecasts given in probabilistic terms were detected depending on gender and age (Handmer and Proudley, 2007). Social and cultural factors may influence the interpretation of the probability of occurrence of a given outcome and the perception of the seriousness of possible non-desirable consequences. Research on cognitive bias and prospect theory (behavioural economic theory that describes the way in which people choose between probabilistic alternatives that involve risk) indicates that people have difficulty in correctly interpreting risk because they are more likely to act to avoid a loss than they are to achieve a gain (Kahneman and Tversky, 1979; Kahneman et al., 1982; Kahneman, 2011). IPBES Deliverable 2a takes this into account when pointing to the fact that the way in which a statement is framed will have an effect on how it is interpreted; for instance, a 10% chance of dying is interpreted more negatively than a 90% chance of surviving. Hence, when assessing and communicating confidence for executive summaries and summaries for policymakers, it recommends considering reciprocal statements to avoid value-laden interpretations. It is advisable that the Task Force on Capacity Building encourages further research on cognitive processes that may help improve the communication of more precise information regarding uncertainties and risks in a probabilistic format.

### 8.4.4 From scenarios to decision making

The process whereby stakeholders engage in a scenario assessment includes the definition of the relevant variables, assumptions, methods and parameterisation, all the way to communicating results, uncertainties and caveats, in the appropriate language and to different audiences (Cash et al., 2003; Folke et al., 2005). There is a variety of science-policy interfaces that enable the two-way communication between scientists and stakeholders needed for a scenario assessment (Chapason and van den Hove, 2009). The most successful of these science-policy interfaces have some institutional way of facilitating or enabling the aforementioned functions over the long periods of time that are often necessary for effective communication. Such institutions have been called boundary or bridging institutions (Cash et al., 2003; Folke et al., 2005; Cash et al., 2006).

The role of bridging institutions in facilitating the science to policy process is crucial, given the multi-scale features of most realistic biodiversity-governance problems, the variety of stakeholders (Section 8.4.1), and the serious problem of communicating the assumptions and the results of 'boundary objects' (Section 8.4.3) such as scenarios. Boundary objects are collaborative products that are both adaptable to different viewpoints, and therefore commonly recognised, and relevant for different actors and robust enough to maintain their identity across these (Clark et al., 2011). In addition to scenarios, other examples of boundary objects are conceptual frameworks, models and reports (Hauck et al., 2014).

Boundary objects resulting from a science to policy process should be communicated actively using the right translation of terms and concepts and, if needed, mediation between stakeholders with different languages, usages and histories (Cash et al., 2003). Such demanding and complicated tasks are better performed institutionally as an institution is more likely than individuals to develop the credibility, memory and experience needed to facilitate the process of developing appropriate boundary objects. Bridging institutions such as IPBES and IPCC can create the conditions not only for the development of boundary objects but also for the uptake of those boundary objects by decision makers and other stakeholders. Bridging institutions can also demonstrate the benefits and use of scenario assessments, so that models and scenarios are more widely used in decision making in a variety of contexts.

In this chapter, we have provided an overview of the multiple aspects of the scenario development cycle and the underlying dialogue between data and model that is amenable for improvement (see summary in Box 8.7). Ultimately, it is up to scientists and all stakeholders to bring these ideas to fruition in order to improve decision-making processes related to the management of biodiversity and ecosystem services.

**Box 8.7:** Summary of key issues to improve scenarios

To increase the uptake of models and scenarios in decision-making processes, assessments should:

- identify key global biodiversity and ecosystem services problems and questions to which they can develop effective and robust answers;

- overcome disciplinary barriers in modelling, data collection, selection and management;
- identify the co-design and co-development of best practices that respond to policy needs;
- define, develop and improve modelling and scenario development methodologies appropriate to the different social contexts and policy needs;
- identify robust model integration and validation techniques that respond to current and future development requirements;
- establish a permanent dialogue between modellers, scenario developers and decision makers to address issues such as common understanding of concepts, transdisciplinarity and infrastructure for resource and knowledge sharing;
- encourage transdisciplinary research leading to a clearer, more effective and broader communication of model and scenario outputs as well as the communication of uncertainties within the cultural context of the human societies involved.

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