8

CHAPTER **8**

IMPROVING THE RIGOUR AND USEFULNESS OF SCENARIOS AND MODELS THROUGH ONGOING EVALUATION AND REFINEMENT

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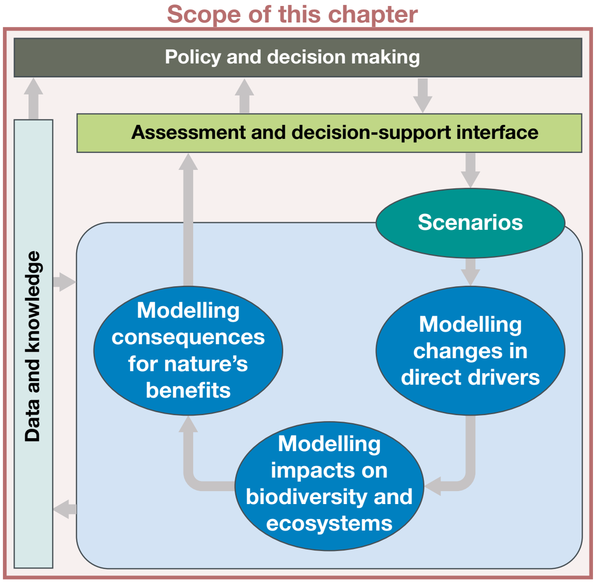
CHAPTER 8

IMPROVING THE RIGOUR AND USEFULNESS OF SCENARIOS AND MODELS THROUGH ONGOING EVALUATION AND REFINEMENT

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.

Scope of this chapter



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 Science-Policy 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.

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 (Diaz *et al.*, 2015).

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).

FIGURE **8.1**

Scenario development and analysis process involving steps (in blue-green circles) such as engaging actors and stakeholders (including ILK), with each step interacting with the data and models (orange arrows) and with information flow between models and data (green 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.

Involvement of stakeholders in scenario development and use

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

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]sCBD, 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.*, 2016; [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]

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).

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; sCBD, 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 Scenarios and Modelling Expert Group, 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 (i) the enhancement of monitoring programmes, (ii) the mobilisation of data, and (iii) modelling for interpolation 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;

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). Desirable 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.

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).

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).

8.2.3 Facilitating data access for model calibration and validation

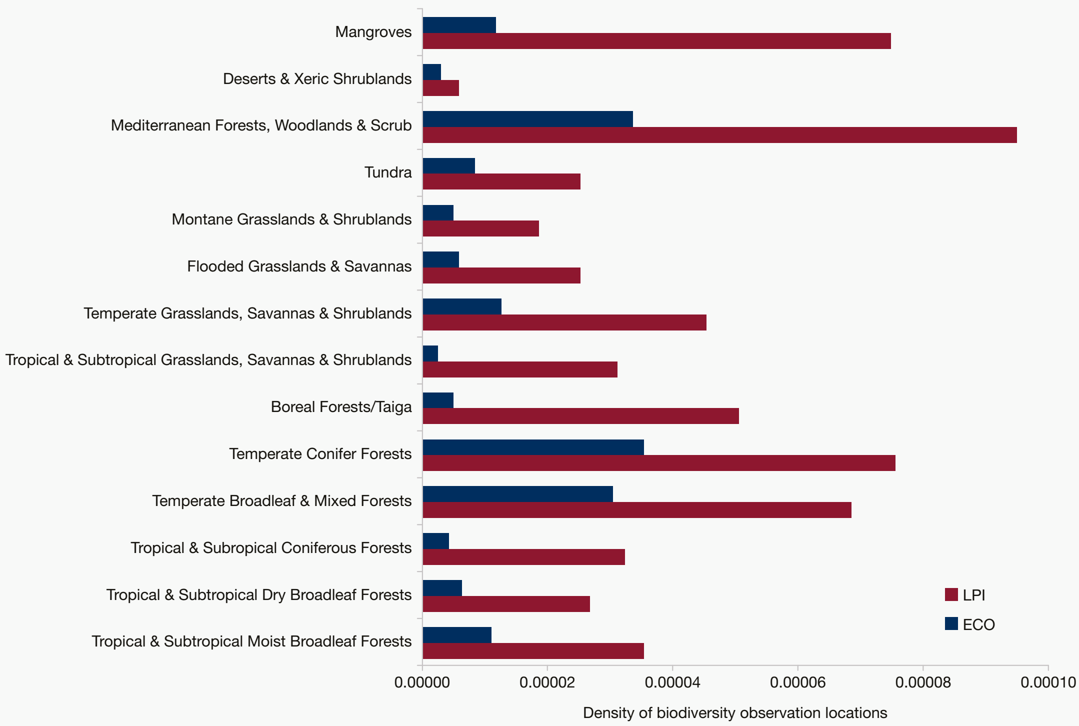
Good practices in sharing data, developing open source databases and platforms, and documenting data access procedures need to be encouraged within the scientific community.

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,

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.



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 publishing 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; 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.teebweb.org/) gives a global overview of the estimates of monetary

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/

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.

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.

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 – https://lta.cr.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

8.3.1.1 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.

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 (https://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.

The goal of the validation: 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.

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.

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

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).

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.

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).

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 5th 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**).

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

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 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 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

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.

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

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 4th 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.

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

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.

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.

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