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

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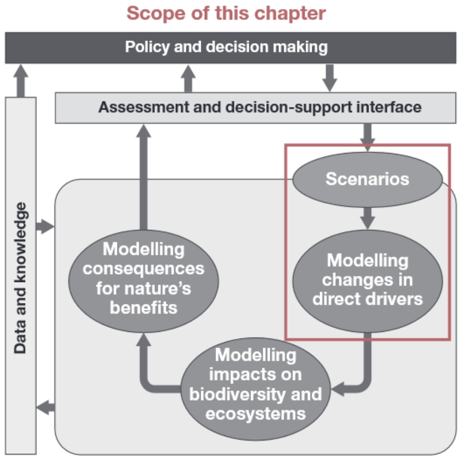
Purpose of this chapter: Provides an overview of broad types of scenarios for addressing the various policy and decision-making contexts introduced in Chapter 2; and critically reviews major sources of scenarios of indirect drivers and approaches to modelling resulting changes in direct drivers that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

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



Key findings

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

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

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

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

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

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

Key recommendations

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

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

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

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

3.1 Introduction

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

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

3.1.1 Definition and classification of direct and indirect drivers

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

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

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

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

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

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

3.1.2 Chapter overview

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

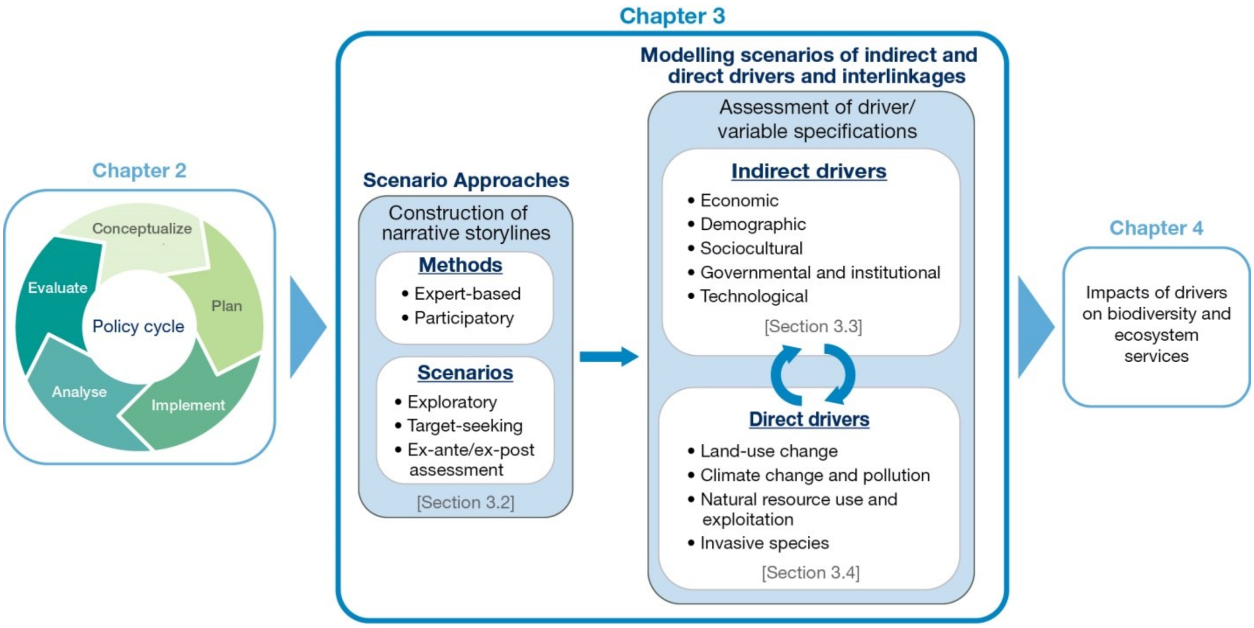


Figure 3.1: Chapter 3 overview**.**

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

3.2 Methodological approaches to scenario and model construction

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

3.2.1 Approaches

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

3.2.1.1 Expert-based approaches

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

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

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

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

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

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

Box 3.1: The **Delphi Technique**

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

3.2.1.2 Participatory approaches

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

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

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

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

3.2.2 Scenarios

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

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

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

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

Table 3.1: Combining scenario approaches and policy objectives.

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

3.2.2.1 Exploratory scenarios

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

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

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

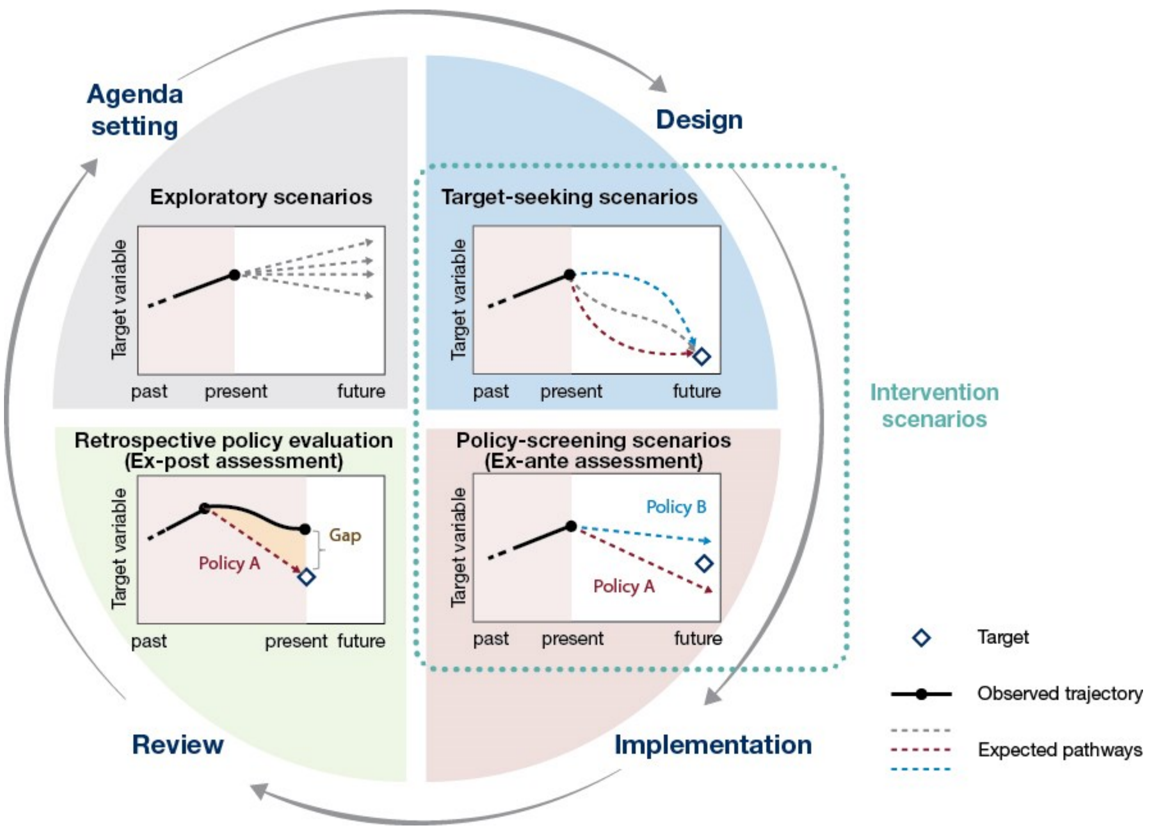


Figure 3.2: Building scenarios of indirect and direct drivers within the policy cycle context for biodiversity and ecosystem services.

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

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

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

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

Main steps for building exploratory scenarios:

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

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

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

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

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

5) Building preliminary scenarios:

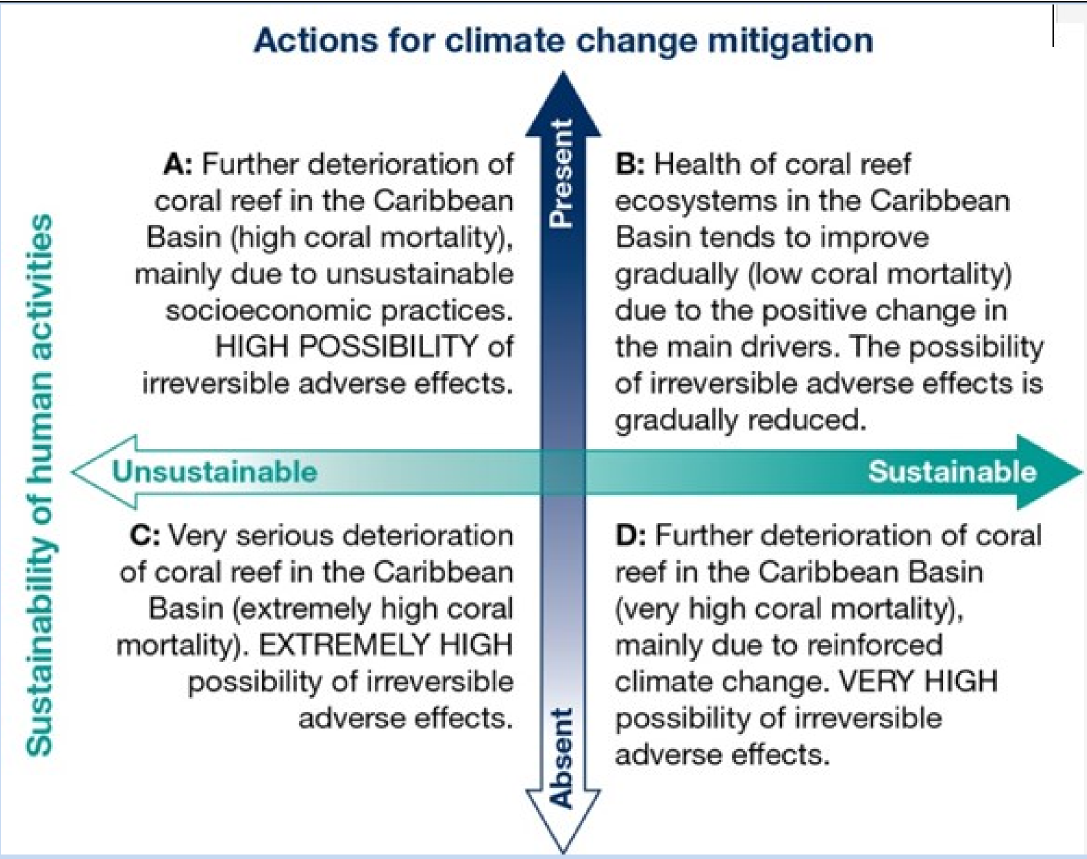


Figure Box 3.2: Scenario matrix

3.2.2.2 Target-seeking scenarios

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

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

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

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

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

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

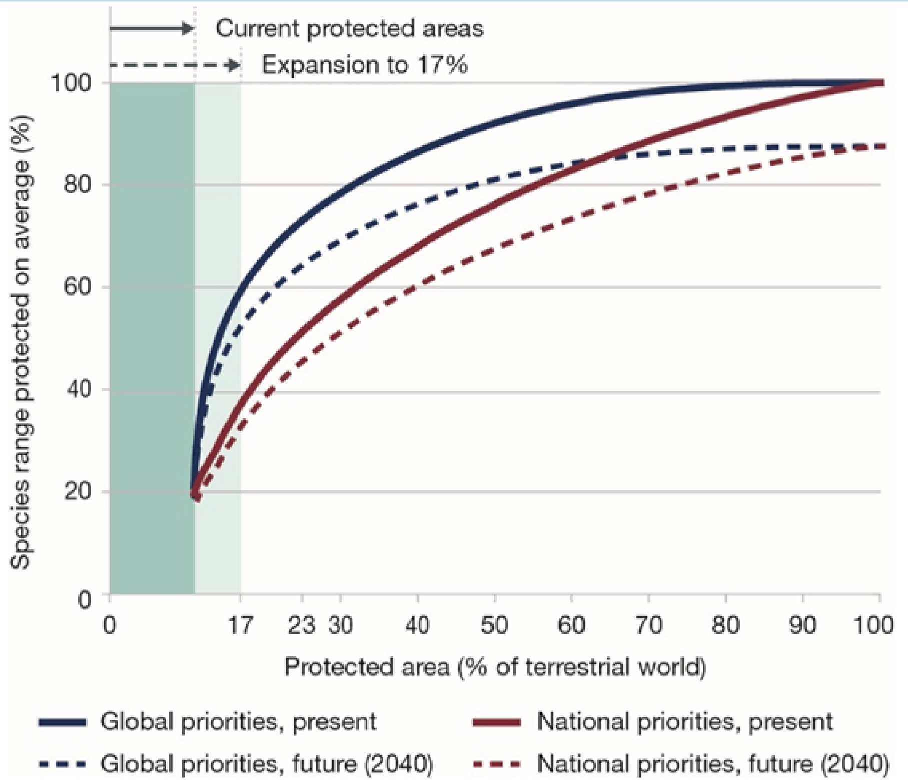


Figure Box 3.3: The relation between the protected area and the maximum attainable protection of species under conditions of the optimum spatial allocation of protected areas. Under global priorities the allocation is globally optimised, while under national priorities the optimisation is based on a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

3.2.2.3 Ex-ante/ex-post assessment

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

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

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

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

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

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

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

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

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

3.2.3 Models

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

3.2.3.1 Modelling methods

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

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

Good modelling practice

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

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

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

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

3.2.3.2 Linking multiple models

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

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

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

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

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

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

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

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

3.3 Scenarios and models of indirect drivers

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

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

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

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

3.3.1 Economic trends

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

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

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

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

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

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

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

3.3.2 Demographic trends

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

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

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

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

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

3.3.3 Society and culture

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

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

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

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

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

3.3.4 Governance and institutions

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

In many countries, factors such as weak governance and institutions, lack of cross-sectoral coordination and illegal activity are cited as key indirect drivers of ecosystem change (Kissinger and Rees, 2010). Common governance challenges include confused goals, conflicts and unrealistic attempts to scale up beyond institutional capacity. Where collective action and conflict resolution mechanisms break down, the governance of ecosystem resources is compromised (Ostrom, 1990). Fragmented legal systems can lead to gaps and conflicts (Techera and Klein, 2011, Pomeroy et al., 2010), while the governance of large-scale ecosystems requires the identification of the heterogeneous, multi-scale and interlinked nature of these systems (Fidelman et al., 2012).

Institutions can promote ecosystem services exploitation. For instance, in Thailand policies that promoted shrimp farming by absentee landlords led to the massive destruction of mangrove ecosystems and thereby the exposure of coastal communities to catastrophic storm and tsunami events (Barbier et al., 2011). Alternatively, public policies can positively affect biodiversity and ecosystem services dynamics as exemplified by recovering fish stocks under the Common Fisheries Policies of the European Union (Fernandes and Cook, 2013). Here, secure private-property rights are widely considered to promote more efficient resource utilisation and property management than open access schemes, although there are many circumstances in which private-property rights do not guarantee resource conservation (Acheson, 2006), in addition to which most common property arrangements involve some degree of private-property management (Ostrom and Hess, 2007). Group size and makeup (e.g. gender) also have important implications for sustainability in situations involving collective resource management (Poteete and Ostrom, 2004; Westermann et al., 2005).

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

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

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

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

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

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

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

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

3.3.5 Technology

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

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

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

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

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

Box 3.5: Bioenergy and indirect land-use change

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

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

3.4 Scenarios and models of direct drivers

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

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

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

3.4.1 Land-use change

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

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

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

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

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

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

Box 3.6: Agroforestry

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

3.4.2 Climate change and pollution

Climate change

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

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

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

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

Box 3.7: IPCC scenarios

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

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

Pollution

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

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

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

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

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

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

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

Box 3.8: Persistent organic pollutants

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

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

3.4.3 Natural resource use and exploitation

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

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

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

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

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

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

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

3.4.4 Invasive species

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

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

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

Box 3.9: Invasive species in the South African context

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

3.5 Lessons learned and the way forward

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

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

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

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

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

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

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

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

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