2. Using scenarios and models to inform decision-making in diverse policy, planning and management contexts

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Key findings

The most appropriate decision support approach and model of biodiversity or ecosystem services to apply in any given decision depends on the decision context. Decisions that impact on biodiversity and ecosystem services are numerous and diverse, and only a small proportion are explicitly considered 'environmental' decisions. Situations in which decision support approaches, including models and scenarios, play a part arise in agenda setting assessments, policy development and implementation, planning, and management. Decision support approaches and assessments often utilize biodiversity and ecosystem service scenarios and models in characterising cause-effect pathways and exploring the consequences of policy, planning and management options. We develop an agenda-setting, policy, planning and management decision typology based on a set of 16 attributes including political scale, ecological complexity, temporal scale, cultural context, complexity of governance arrangements, and types of uncertainty considered. We attempt to identify decision support and assessment approaches relevant to characteristic combinations of attributes in our typology.

We identify seven key barriers that impede the widespread and productive use of models and scenarios in agenda setting, policy, planning and management. These range from a lack of appreciation among decision makers about the potential benefits of using models and scenarios, to a lack of willingness on the part of modellers to properly engage in real-world decision-making and undertake relevant analyses. A huge number of agenda-setting and decision support approaches now exist that have been used in a wide variety of policy contexts, utilizing a wide variety of biodiversity and ecosystem service models and scenarios. On reviewing a large number of assessment and decision processes, we find that a key ingredient for the successful application of structured decision support, models and scenarios is the dedication and continuity of involvement of decision support facilitators and modellers in close collaboration with decision makers and decision making processes. We find that the primary impediments to the widespread and productive use of models and scenarios in policy, planning and management is a general lack of trust in modellers, models and scenarios, a lack of understanding among decision makers about the positive role that models and scenarios can play, a general lack of decision support, modelling and scenario analysis skills relative to the number

of policy, planning and management processes, a lack of data to underpin the models and scenarios of most interest to policy makers and managers, a lack of willingness on the part of modellers to engage fully in real-world decision problem and develop the most relevant scenarios and models for the problem at hand, a lack of transparency in approaches to modelling and scenario development, and complex political agendas that are not amenable to the transparency ideally associated with good modelling and scenario analysis. Increased collaboration between modellers and decision makers will lead to increased trust, better and more relevant models and scenarios, and a culture of decision support based on models and scenarios that is robust to complex political agendas.

The spatial and temporal scales of many biodiversity and ecosystem services models and scenarios have to be improved to better match not only jurisdictional requirements and decision time frames but also the cross-scale linkages among decision makers and their stakeholders. When designing models and scenarios for the purpose of assessment and decision support, the needs of policy and decision makers should be paramount in determining data needs, necessary model outputs, and consequently, the types of scenarios and models that are developed. Significant progress has been achieved in understanding impacts and feedbacks of state-variables across various organizational levels (i.e. genetic, species, ecosystem and landscape) using models and scenarios that are dynamic in space and time. Recognising the complexity and stochasticity of socio-ecological systems, knowledge about the state of key biodiversity and ecosystem service variables and how socio-ecological systems function and respond to stressors and human interventions should be constantly enhanced through collection of new data at multiple organization levels and by monitoring the impacts of decisions. Assessment and decision support needs should drive data collection priorities, and the choice of scenarios, models and model outputs.

Uncertainties in decision making process can have negative social, economic and ecological implications and need to be identified and addressed through the decision making process at the policy, planning and management level. Environmental problems and the process of finding technical and management solutions to these are challenged by stochastic, linguistic, scientific and decision uncertainties with various levels of complexity and reducibility. Uncertainties that are very complex, have low reducibility and entail many trade-offs can be successfully dealt with through deliberation (not just communication) that allows feedback and learning among decision-makers and stakeholders.

The issues of scale serve as impediments to integrating indigenous and local knowledge systems in models and scenarios and improving participation of indigenous and local people in the biodiversity and ecosystem services decision making process. Ecological systems are complex and difficult to capture by only one scientific discipline or knowledge tradition. The livelihoods of traditional knowledge holders are highly dependent on biodiversity and ecosystem services but they are frequently excluded from policy decisions, particularly above the national level. In order to make use of indigenous and local knowledge systems and encourage greater participation, issues of scale must be resolved to allow the integration of local knowledge with regional or global level analysis.

Key recommendations

We recommend that the thematic (IPBES deliverable 3b), regional (2b) and global (2c) assessments as well as the policy and decision tools catalogue (4c) take into consideration the agenda-setting, policy, planning and management typology provided here when choosing the decision support approach most relevant to their decision or assessment context.

We recommend that the task force on Capacity Building (1a & b) develop capacity in modelling and scenario analyses within the regional and global assessments for both facilitators and policy modellers.

We recommend that the modelling and scenarios expert group (3c) look at the decision time frames of relevance to decision makers and their stakeholders and the policy and decision tools catalogue (4c) to consider jurisdictional needs and decision time frames in developing the catalogue. We recommend that IPBES task force on Knowledge, Information and Data engage with funding agencies to collect data targeted towards decision-making needs, at multiple organisation levels and to monitor impacts of decisions on composition, structure and function of biodiversity and ecosystem services.

We recommend that the thematic, regional and global assessments identify the capacity needs for dealing with scientific uncertainties and that the task force on Capacity Building develop capacity for communicating different types of uncertainties to the decision makers through deliberation.

We recommend that thematic, regional and global assessments, in close cooperation with the IPBES Task Force on Indigenous and Local Knowledge (1c), use methods that can integrate different scales and recognize the importance of mutiple and diverse knowledge systems.

2.1. Introduction

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

2.1.1. Overview - agenda setting, policy, decisions making, and knowledge needs

Within the relatively small scope of decisions that are made with regard to biodiversity and ecosystem services, three broad types of decision making context can be identified; policy development, planning, and management. A ***policy*** is a principle, usually stemming from an articulation of a group or individual's values that guide actions. For example, governments and large public or private organizations commonly design policies to encourage economic prosperity or wellbeing that have implications for biodiversity and ecosystem services. Equally, policies can be developed specifically to address biodiversity or ecosystem service issues or problems that may have impacts on other values such as opportunities for wealth generation. Decision support protocols, underpinned by biodiversity and ecosystem service models or scenarios have been used to support policy development and implementation. A key component of policy development is policy ***agenda setting***. This is an area of policy development in which scenarios and models have played a key role (e.g. MEA 2005, Alkemade et al. 2009, Leadley et al. 2014). ***Planning*** is the process of prioritizing and scheduling actions in order to achieve individual or organizational goals. For example, land-use planning can be undertaken to identify which activities will be allowed or encouraged in particular parts of the landscape in order to achieve landscape-level objectives for a range of criteria such as agricultural productivity, tourism service provision, and biodiversity conservation (Young 1993). Formal decision protocols and models of biodiversity and ecosystem services have been used to support land-use planning at multiple planning scales (SAPM 2009). ***Management*** decision-making occurs in such a wide variety of management contexts, at every spatial and temporal scale that a formal definition of management decision-making seems trivial. In complex instances of management decision making that involve highly uncertain benefits and costs due to complex ecological or social system dynamics, and multiple criteria for measuring success, formal decision protocols can be extremely useful for ensuring that management is effective and efficient in meeting objectives for biodiversity, ecosystem service, and other criteria (Runge et al. 2011a).

Policy, planning and decision-making processes have specific knowledge needs that can, and arguably should be partly met by biodiversity and ecosystem service models implemented under exploratory and intervention scenarios (see Figure # in Chapter 1). This chapter sets the policy and decision-making scene for the three other chapters of this deliverable that provide more detail on modelling and scenario development approaches (Chapters 3, 4, and 5). It links to Chapter 3 by identifying types of scenarios required to underpin decision making, and to chapter 5 by identifying the role of model outputs in agenda-setting, policy and management. This chapter provides the foundation for Chapters 6 and 7 by highlighting scales and domain over which types of decisions occur, and capacity building needs in the area of model-supported decision analysis. A view to the future of agenda setting and decision-making offers an entre to Chapter 8 that highlights future developments, which may see increased used of models and scenarios in decision-making.

2.1.2. Audience and task forces

This chapter aims to inform readers about the possibilities and opportunities for using models and scenarios in a structured way for agenda setting and to inform decision-making. We principally

address the other IPBES work programmes, regional and global assessment (Del.2b&c), other thematic assessments (Del. 3b), the scenarios and models task force (TF3c) and the deliverable on policy and decision tools (Del.4c) which will develop an online catalogue of policy support tools and methodologies relevant to IPBES related activities. However, we also address deliverables 1a/b on capacity building, 1c on indigenous and local knowledge, 1d/4b on knowledge information and data, 3d on evaluation and 4d on stakeholder engagement.

2.2. Policy and decision making context

2.2.1. The policy cycle

The policy making process is often studied as a sequence of stages to organize and systemize the development and implementation of public policy (Figure 1). The advantage of frameworks is that they can help standardize and communicate best practice, often focussing on generic features rather than on specific contexts. Although empirical evidence shows that real world decision-making usually does not follow the idealistic sequence of discrete stages (Jann and Wegrich (2007), the policy cycle is particularly useful for us here to help organize our discussion of the role of scenarios, models, and decision support approaches in decision-making. A plethora of published frameworks exist that describe very similar steps and approaches for structuring and implementing policy and decision-making under uncertainty and complexity. In most representations three broad phases consistently appear; (i) review (evaluation) and agenda setting, (ii) policy design, and (iii) policy implementation. Our use of this particular representation of the policy cycle is not intended as a judgement on the relative value of this or other representations.

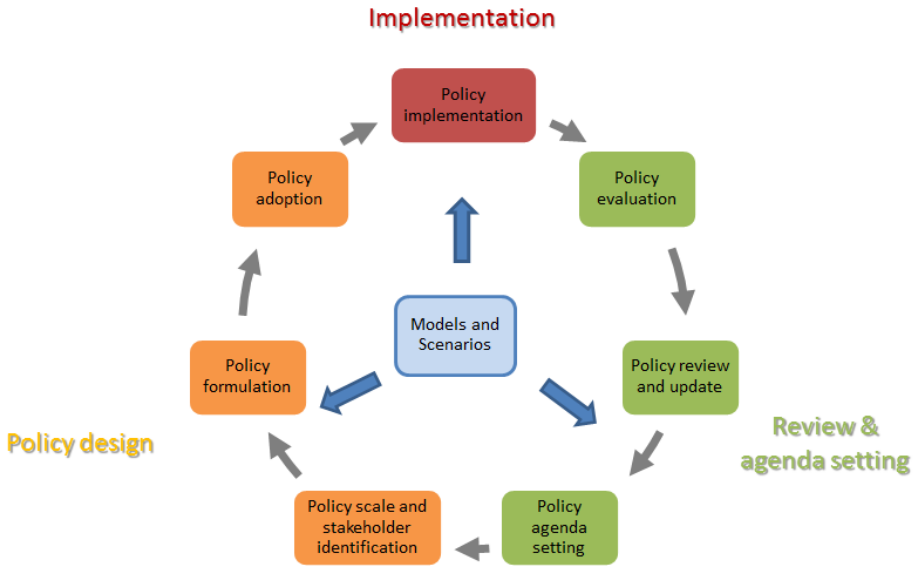

Figure 2.1.: A theoretical framework for policy formulation, implementation and evaluation (adapted from Howlett et al. 2009) identifying the activities most likely to utilize models and scenarios. The policy cycle is frequently described as iterative and similar to adaptive planning and management (McFadden et al. 2011, Walters 1986).

2.2.2. Attributes that define decision context

Almost every policy, plan and action in every sector from health to manufacturing, at every spatial and organizational scale from the individual to the globe impacts in some way on biodiversity and ecosystem services. The number and types of decisions made appear to defy classification; and are practically infinite (Fisher et al. 2009). The bulk of decisions or choices made on a daily basis that impact most on biodiversity and ecosystem services are seldom described or even conceived of as *environmental* decisions. Almost all of them are undertaken by people outside the environment sector with little or no consultation with environmental professionals. It is our goal to broadly categorize decisions that impact on biodiversity and ecosystem services according to key attributes that define a decision context. The point of this exercise is to try and reduce some of the complexity and confusion about the range of decision protocols that helps support decisions that impact on biodiversity and ecosystem services and the contexts in which they might be most fruitfully applied. Having mapped out the decision space and scoped some potentially useful decision support approaches, we then hope to draw a link between the decision contexts, the relevant tools and the role that models and scenarios can play.

Most environmental decisions are characterized by high ***uncertainty*** due to a lack of knowledge about cause-and-effect relationships (Ludwig et al. 2001), stochasticity, and a variety of other sources. The degree and type of uncertainty inherent in a particular decision problem determines the sorts of analytical and decisions support approaches that can be applied (Peterson et al. 2003, Regan et al. 2005) and partly motivate the need for models and scenarios. We deal with the role and implication of uncertainty in decision-making, modelling and scenarios in section 2.3.3.

Decisions that impact on biodiversity and ecosystem services take place across many ***spatial and temporal scales, jurisdictions and administrative contexts***; from the largest most complex systems involving many actors, cultural perspectives, values and services, to problems involving relatively few actors and species, and services at the local level (Table 1). The scale of the policies, plans and actions will vary depending on the relevant drivers of change, both direct and indirect. Matching the response to the scale of the problem and drivers and ensuring that multiple responses do not create conflict can be a huge challenge. Biodiversity and ecosystem services have specific spatial and temporal distributions that overlap with human management units or jurisdictions in complex ways. Various stakeholders have rights, obligations and interests at a variety of spatial scales. Global responses to ecosystem problems are warranted when those problems potentially affect all people and ecosystems of the world, in other words common pool resources. Multilateral, regional and bilateral agreements require consensus by a group of nations but implementation often requires action within national boundaries. National policies exist independently of agreements with other nations, highlighting the problem of policies and plan that conflict across scales. The scale at which human and biotic processes operate influences the sorts of decision approaches, models and scenarios relevant to a particular decision. Spatial scale partly determines who will be represented in a decision problem and whose interests are considered.

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

Axes	How each varies (categories or continua)		
Political scale	Local	->	Regional
Policy cycle stage	Review, agenda-setting, design, implementation		
Type of uncertainty	Stochastic, epistemic, linguistic, decision		
Governance attributes			
Decision making entity	Public, Private		
Decision process	Participatory		Top down
Level of governance	Single		Multiple
Sectors	Single		Multiple
Temporal scope	Seconds		Millennia
Stakeholders	Single		Multiple
Objectives	Single		Multiple
Cultural context (value/knowledge system)	Homogenous		Diverse
Biophysical attributes			
Ecological complexity	Single ecosystem		Linked, multiple systems
Landscape complexity	Single species		Multi species
Temporal extent	Short term		Long term
Temporal resolution	Seconds		Millennia
Spatial extent	Local		Global
Spatial resolution	Metres		Kilometres

Many environmental problems are characterised by high ***social and cultural complexity***, leading to divergent views about the appropriateness of policies, plans or actions and criteria for what constitutes a good outcome. Differences in capacities and power determine the effectiveness of stakeholder representation and the acceptability of decision outcomes. Large and wealthy organizations, including companies and national governments may have greater resources and better access to information than other stakeholders, leading to a greater influence over the decision process. Assessing the impacts on livelihoods of policies, plans and management options may require culturally specific, local level understanding to properly evaluate costs and benefits to all stakeholders (Runge et al. 2011a, Nordström et al. 2010, Rowland et al. 2014). Cultural norms, values, practices, ideologies and customs shape people's understanding of their needs, rights, roles, possibilities and hence on their actions, including engagement in policy development and implementation (Borrini-Feyerabend et al. 2004). All stakeholders use their beliefs as the basis for determining the range of

options they will consider and the criteria by which they will measure outcomes. The importance of taking into account multiple belief systems during policy formulation is being increasingly recognized especially in areas where indigenous people have consolidated their property and representation rights (Tauli-Corpuz et al. 2010, UNDRIP 2007, Runge et al. 2011a).

Some 'local' decisions take place within a particular ***ecosystem or geographic domain*** that can be considered for the purposes of the decision process discrete and sufficiently buffered from the ecological processes playing out in other systems, so as to simplify the characterisation of biodiversity and ecosystem service values and dynamics. However, many land-use planning and policy processes play out over multiple ecosystems that are connected by complex flows of biotic and abiotic resources, and which are subject to multiple different types of ecological and social dynamics that may play out of multiple temporal scales. For example, some integrated catchment management strategies must consider simultaneously terrestrial, river, estuarine and near-shore ocean ecosystems, each with unique economic drivers and pressures such as agriculture, aquaculture, and fishing (e.g. Brodie et al. 2012).

The ***governance system*** under which decisions are made, and the degree to which power over a given decision is shared among actors contribute significantly to the nature of the decision approach taken and the types of decision support, models and scenarios likely to be relevant. For example, so-called 'top-down', 'single-actor' decision problems may be amenable to application of economic optimization approaches, while more 'participatory', 'multi-actor' decision structures may be much less amenable to such approaches. Sequential decision processes provide the opportunity to value the role of learning and to establish formal programs of 'continuous improvement', often invoking ideas embodied in adaptive management (Walters 1986). However, with this opportunity comes complexity. Many reasons have been proposed for the auspicious lack of working examples of adaptive management (AM) in broad scale, multi-objective decision problems (Walters 2007, Wintle and Lindenmayer 2008). The complexity associated with achieving a working adaptive management process in all but the simplest of resource management problems may be partly to blame.

In the remainder of this chapter we will explore how the attributes of decision context influence the choice of decision analysis and support approach that is most appropriate and how this, in turn influences the types of models and scenarios that are used.

2.3. Assessment and decision support

Many methods, approaches and tools exist to support of environmental policy, planning and management decisions ranging from specific technical tools within a very specific domain of application such as mathematical optimization approaches, through to broad frameworks such as 'structured decision making' (*sensu* Gregory et al. 2012) and adaptive management (sensu Walters and Holling 1983) that set out approaches for dealing with most challenges confronting practitioners charged with managing a complex decision problem. In many instances, the application of more general approaches to decision support includes application of some specific tools such as models and scenarios. For example, it is very common for a 'structured decision making' exercise to include some

multi-criteria analysis (*sensu* Hajkowicz et al. 2000, Hajkowicz 2008), perhaps supported by some economic cost-benefit analysis. Here we provide a non-exhaustive overview of the main families of environmental decision support approaches in a rough order of most specific to most general. We start with an overview of the attributes of each family of methods and approaches, and summarize a single case study application of a particualr approach within each family. We provide a table documenting how each approach fits within our coarse decision-context typology, which we expand on in Appendix #.

2.3.1. Assessment and policy agenda setting

Agenda setting is one of three broad phases in the policy cycle. Arguably, it is the most important because it motivates and sets the direction for policy development and implementation. Models, scenarios and decision support analyses can all play a role in agenda setting. Numerous important examples exist. The first Global Biodiversity Outlook (Secretariat of the Convention on Biological Diversity 2001) presented information from national reports and a global evaluation of biodiversity trends (WCMC 1992). These analyses were later augmented with exploratory scenario analysis in the second Global Biodiversity Outlook – the *Crossroads of Life on Earth* study (Secretariat of the Convention on Biological Diversity and Netherlands Environmental Assessment Agency 2007). This study used GLOBIO as a modeling framework to assess the impact of environmental drivers on biodiversity and explore policy options in the form of intervention scenarios to reduce biodiversity loss and achieve the 2010 targets for biodiversity. The third Global Biodiversity Outlook (Secretariat of the Convention on Biological Diversity 2010) also presents biodiversity scenarios and tipping points contained in a study incorporating the results of the Millennium Ecosystem Assessment (MEA), the Global Biodiversity Outlook 2 and the Global Environmental Outlook 4, as well as the Mini Climate Assessment Model (MiniCAM) (Leadley and Pereira 2010). The fourth Global Biodiversity Outlook provides a mid-term assessment of progress towards the implementation of the *Strategic Plan for Biodiversity* and achievement of the Aichi Targets (Alkemade et al 2009, Leadley 2014). Each of these assessments have contributed significantly to the current policy agenda pertaining to biodiversity and ecosystem services at multiple spatial scales across multiple jurisdictions (Figure 2.2.).

Global agendas play out at regional and national scales in many ways. Referring directly to the CBD, the National Performance Assessment and Sub regional Strategic Environment Framework for the Greater Mekong Sub region (GMS; ADB 2006) was developed to guide the GMS Core Environment Program (CEP), through which the GMS governments create a vision and framework for long term investment in environmental governance and institution building, and environmental protection in the main development sectors, and conserving biodiversity. In this process, the GLOBIO3 model underpinned assessment of different policy options to reach biodiversity targets in the region (Figure 2.2.).

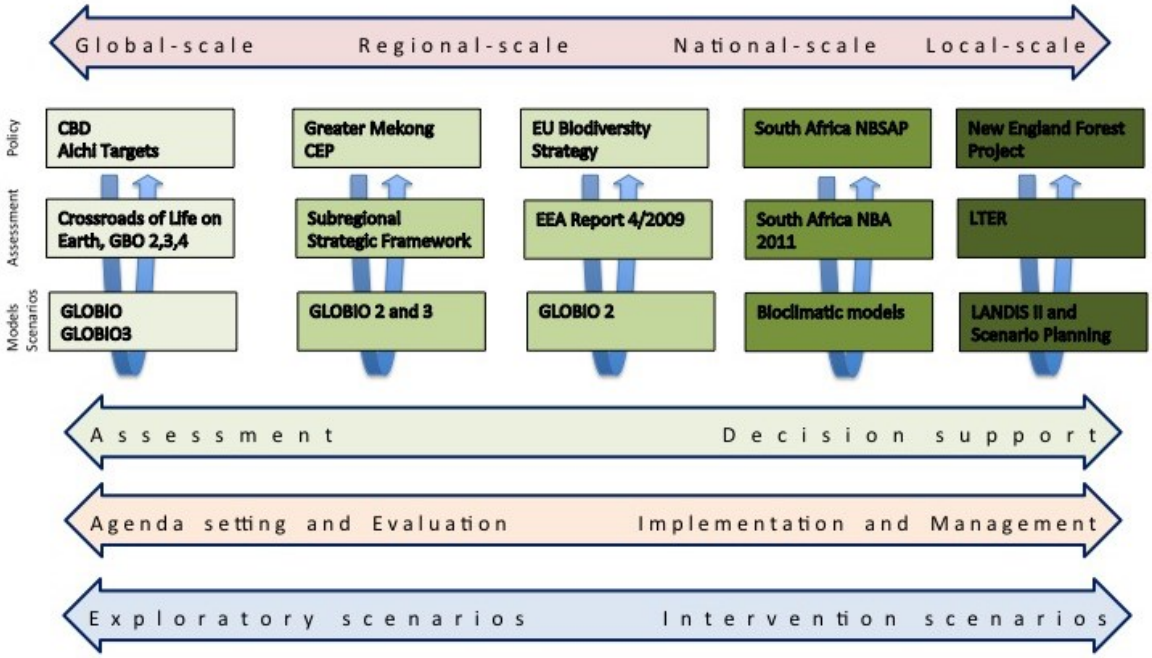

Figure 2.2.: The relationship between spatial scales, phase in the policy cycle and scenario type with Aichi targets and subordinate activities as an example. We recognize that there is not one-to-one correspondence between spatial scale and policy cycle phase or scenario type, however, this scheme does provide some insight to commonly observed hierarchies of policy, planning and action and some of the tools that used at different levels in the hierarchy.

At a regional level, the European Commission developed the EU Biodiversity Strategy (European Commission 2011) which was informed by an assessment of the 2010 biodiversity targets (EEA 2009). These activities represent both policy formulation and evaluation, following from the agenda set by CBD and MEA (Figure 2). International fisheries policy in the same region has been influenced by model and scenario assessment at the same scale (Box 2.1).

Box 2.1. Models and scenarios for agenda-setting, policy and regulation at regional scale: European international fisheries policy.

The European marine policy frameworks have adopted ecosystem-based management, which requires indicators that describe pressures affecting the ecosystem, the state of the ecosystem, and the response of managers (Jennings, 2005). This adoption of ecosystem based management is due to a shift in research effort from single species to ecosystem-based concerns which reflects a growing recognition that an ecosystem approach may help to underpin improved management (Jennings 2004).

Numerous published models describing the complexity of marine ecosystems (Baird et al. 1991; Baird and Milne 1981; Baird and Ulanowicz 1989) underpin indicators that drive the Marine Strategy Framework Directive (MSFD; 2008/56/EC). The MSFD requires that European Union Member States achieve “Good Environmental Status” (GES) under 11 descriptors of the marine environment by 2020.

Of these 11, descriptor 4 (D4) addresses marine food webs: "All elements of the marine food webs, to the extent that they are known, occur at normal abundance and diversity and levels capable of ensuring the long-term abundance of the species and the retention of their full reproductive capacity". The D4 indicator stipulated in the Commission Decision (European Commission, 2010; 2010/477/EU; Rogers et al., 2010), addresses three criteria related to food web structure and energy transfer. Descriptor 1 on biodiversity also relates to species distribution ranges, habitat extent, habitat condition and ecosystem structure. All of these measures are dependent on habitat and ecosystem models as none are directly measurable at broad scales in the marine environment.

Policy implementation at the national or subnational level often requires spatially explicit analysis to identify understand, prioritize and manage the factors that influence environmental outcomes. The use of national scale input data and refinement of locally relevant scenarios allows local assessment to inform local policy options. At the national and subnational scale there exists a multitude of methods for exploring, evaluating and decision-making in socio-ecological systems (see following sections). The South Africa National Biodiversity Strategy and Action Plan (DEAT 2005) guides policy design and implementation at finer scales and was informed by the National Biodiversity Assessment (Driver et al 2012) that used bioclimatic models to incorporate climate resilience into species and ecosystem planning. In New England, USA the Harvard Long Term Ecological Research Program (LTER: Foster et al 2010) used scenario planning and LANDIS-II landscape simulation modeling to inform land-use policy and conservation decisions (Figure 2.2.; Thompson et al 2014).

2.3.2. Families of decision support tools

Methods and approaches to support policy, planning, management decisions might sensibly be considered as existing within a continuous, multi-dimensional space representing decision context defined by the attributes listed in table 1. For the sake of presentation we categorize methods into five broad families that we briefly describe here with the help of case studies.

2.3.2.1 Multi-objective approaches to analyze trade-offs

Risk analysis models motivate decision-making on the basis of *expected* consequences. That is, the calculation of risk as the product of likelihood and consequence is essentially an estimate of expected (dis)utility (Savage 1954). While consideration of adverse consequences alone will often suggest the desirability of avoidance or mitigation measures, conditioning estimates of consequence with assessment of likelihood may imply that such measures are not warranted. If estimates of likelihood and consequence are unbiased, then decisions based on risk should lead to more effective allocation of resources (Arrow and Lind 1976).

The real world challenges of decision-making are seldom so simple. Consequences are not restricted to impacts that can naturally or readily be described in single criterion (e.g. monetary) terms. Multiple values imply multiple objectives each requiring estimates of expected consequence. Uncertainty about consequences and likelihoods brings into play complex risk preferences that must be considered. All formally considered decisions involve alternatives and cause-and-effect predictions of expected consequence. When predictions are made over multiple objectives, an additional element is required to resolve the decision problem: the articulation of preferences or trade-offs reflecting the

relative importance of the different objectives (Howard 2007). Most environmental policy, planning and management decisions involve trade-offs (Keeney 2002).

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

Consequence tables are the first of the multi-objective decision support tools described here to deal explicitly with trade-offs. There are three core elements to any multi-objective decision problem; alternatives, expected consequences, and trade-offs. These elements are compactly reported in a consequence table. An example is shown below (Table 2), where alternatives comprise six hypothetical candidate options for reducing impacts on a near-shore reef system resulting from nutrient outflow from an agricultural catchment. The table can be populated with qualitative or quantitative estimates of expected consequence. Experts and non-expert stakeholders alike are notoriously deficient in their capacity to make internally consistent probabilistic judgments (Hastie and Dawes 2010). Modelling tools that assist in the coherent treatment of probabilities include fault tree analysis, event tree analysis, Markov analysis, Monte Carlo simulation and Bayes nets. For example, Jellinek et al. (2014) developed a Bayes net to explore the relative improvement in vegetation condition resulting from a range of investments in woodland management including reducing stock grazing and direct restoration activities.

Table 2.2.: The example below uses coarse verbal (negative) impact descriptors typically seen in a qualitative risk matrix approach. Trade-offs involve consideration of the performance of each alternative against each objective.

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

The preparation of a consequence table itself offers substantial insulation against the pitfalls of unaided decision-making. But unless the decision problem can be meaningfully simplified to two or three objectives and two or three alternatives, the cognitive and emotional demands on decision-makers and stakeholders can lead to poor outcomes. In many instances, a consequence table can be simplified through identification of the *strictly* non-dominated set of alternatives, and consideration of *practically* dominated alternatives and redundant objectives. An alternative is *strictly* dominated if, in comparison with any other single alternative, it performs worse on at least one objective and no better on any other objective. Driscoll et al. (2010) illustrate identification of the non-dominated set in a hypothetical trade-off between asset protection and biodiversity conservation in the context of wildfire management.

If all expected consequences can be assigned a monetary value then ***benefit-cost analysis*** may be applicable. Selection of the alternative with the highest benefit-cost ratio has a strong basis in public policy and welfare economics. However, the monetisation of non-market impacts is difficult. Where revealed preferences are deemed inadequate or absent, robust techniques for stated preference are available (Bennett and Blamey 2001), but the time and resources required to apply these methods are substantial. In any case, stakeholders are unlikely to feel comfortable with monetisation of all objectives, especially those dealing with social and environmental outcomes. In addition, the impact of discounting over time becomes more difficult when dealing with future scenarios.

Maguire (2004) cites two interacting flaws commonly encountered in risk-based decision support: (a) incoherent treatment of the essential connections between social values and the scientific knowledge necessary to predict the likely impacts of management actions, and (b) relying on expert judgment about risk framed in qualitative and value-laden terms, inadvertently mixing the expert's judgment about what is likely to happen with personal or political preferences. The family of techniques under the banner of ***Multi Criteria Decision Analysis (MCDA)*** seek to avoid these flaws through explicit separation of the task of causal judgment from the task of articulating value judgments or trade-offs (Ananda and Herath 2009). See example in Box 2.

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

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

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

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

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

AHP has also been criticised on theoretical grounds because it allows rank reversal upon introduction of a new alternative - a violation of decision theory's independence of irrelevant alternatives axiom (von Neumann and Morgenstern 1944). The modified AHP (mAHP) is free of this problem. It uses standard MAVT techniques to obtain marginal value functions, and limits the use of pairwise comparisons to the derivation of weights. Moffett and Sarkat (2006) advocate use of mAHP because of the relative ease of obtaining weights. But like direct weighting, weights obtained through pairwise comparisons via mAHP result in poor capture of stakeholder preferences. In general respondents tend to assign weights according to the perceived importance of objectives, irrespective of the consequences associated with the specific alternatives being considered. Any weighting technique that fails to promote normative interpretation of weights through explicit consideration of the range of consequences is inadequate (Steele et al. 2009).

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

There are numerous techniques and software packages that fall under the banner of outranking (e.g. ELECTRE, PROMETHEE, GAIA; see Figueira et al. 2005 for details). The techniques vary according to how expected consequences are characterised. If a consequence table is populated using qualitative ordinal descriptors of impact (as in section 2.3) ELECTRE can informally assist stakeholders progress trade-offs and difficult decisions involving more than a handful of objectives and alternatives. While

other outranking techniques can be used where consequence estimates are quantitative or semi-quantitative, there is little argument for doing so, because in these circumstances MAVT offers a much firmer normative basis for decision-making.

The formal description of ***Multi-attribute utility theory (MAUT)*** developed by von Neumann and Morgenstern (1944) remains a high point in the theory of multi-criteria decision-analysis. It is also a wholly impractical approach to typical multi-objective, multi-stakeholder problems. Many of the developments and refinements of Multi-criteria decision analysis since the 1950s are essentially pragmatic short cuts for MAUT. MAUT can be used when a consequence table is populated by statistical distributions describing probabilistic uncertainty in the performance of each alternative against each objective. These circumstances are rare indeed, especially in natural resource management. Aside from difficulties in obtaining detailed probabilistic causal judgments, there are distinctly onerous demands on decision-makers and stakeholders in the elicitation of trade-offs under MAUT. Populating a consequence table with probabilistic outcomes clearly defines a strong role for models and scenarios. In practice, only the most committed and indefatigable participants in group decision-making settings are capable of formally addressing trade-offs using MAUT.

Box 2.2. Multi-criteria decision analysis case study – the use of a web-based Multi-criteria decision analysis system in participatory environmental decision making in Finland

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

2.3.2.2. Optimization approaches

There are potentially thousands of alternative options in most realistic planning and management decision problems. Various mathematical programming techniques from the field of Operations Research are available to help identify better (or best) candidates from a large set.

Linear Programming (LP) and ***Stochastic Dynamic Programming*** (SDP) employ algorithms designed to optimise an objective function under specified constraints (Chankong and Haimes 2008). In LP, a static linear relationship (or near-linear) relationship between actions and expected consequences is required. This may be inappropriate in many ecosystems, where outcomes for objectives are dynamic and non-linear in relation to actions or sets of actions. With detailed understanding of cause-and-effect, SDP can accommodate non-linear, dynamic outcomes associated with the stochastic risk (e.g. risks associated with wildfires) superimposed on the deterministic influence of management actions (e.g. fuel reduction burning in high fire risk places). SDP recognises that what might be considered a desirable action depends on the state of the system (Minas et al. 2012, Richards et al. 1999). For example, low fuel loads and a forest age structure skewed towards regrowth may imply lesser need for burning compared to circumstances where fuel loads are high and old growth is relatively well represented. The capacity to capture greater realism in SDP is attractive, but computational overheads and the requirement for sophisticated causal understanding mean that most applications are substantially simplified. ***Goal programming*** (GP) avoids the naive binary logic of a step function that is commonly employed in setting objectives for LP and SDP. GP requires specification of a performance aspiration for each objective. The underlying algorithm searches among the candidates for the alternative having the minimum multi-dimensional distance to the goal set (Chankong and Haimes 2008). A single decision-maker can use the method profitably. In a multi-stakeholder setting, GP is open to abuse because stakeholders will tend to manipulate outcomes through articulation of insincere positions on what might be considered an appropriate goal for each objective.

2.3.1.3 Scenario-based approaches

Scenarios are plausible and internally consistent, but not necessarily probable, futures (Schwartz 1996). Börjeson et al. (2006) refer to scenario planning as a tool for exploring possible, probable and/or preferable futures. Scenarios can be divided into three main categories; these are predictive which tries to respond the question *"what will happen*?", explorative which tries to respond "*what can happen?"* and normative scenarios "*How can a specific target be reached?*". Unlike forecasting, the focus of ***scenario planning*** is not to assess the probability of future events, rather, it explores possible futures that may arise under different conditions, and what those different futures might mean for current decisions (Schoemaker 1995). Assumptions about future events or trends are questioned, and uncertainties are made explicit (Bohensky et al. 2006). Scenario planning typically takes place in a workshop setting. Participants explore current trends, drivers of change, key uncertainties, and how these factors might interact to influence the future (Schoemaker 1993). To do so, they draw on both qualitative and quantitative information, including datasets (WCS & Bio-Era 2007), spatially explicit data (Santelmann et al. 2004), and expert/stakeholder judgement (Schoemaker 1993). Based on this information, a set of plausible future scenarios is developed. Participants then consider a range of policy or response options, and assess how robust those options are to the different scenarios developed (See example of scenario planning in Box 3).

One of the early applications of scenario planning was to navigate an oil crisis in the 1970s. Shell Oil identified a plausible scenario – considered unlikely and rather radical – where a coalition of oil exporting countries limited production and drove prices up (Peterson et al. 2003). The company adjusted its business practices to buffer itself against this scenario, by making its shipping and refining processes more efficient. In recent years, there have been many applications of scenario analysis on a landscape scale (Steinitz et al. 2003, Baker et al. 2004, Berger and Bolte 2004, Hulse et al. 2004, Santelmann et al. 2004, Shearer 2005, Walz et al. 2007). Many studies have defined landscape scale scenarios using qualitative techniques based on participation of stakeholder groups (Hulse et al. 2004; Patel et al. 2007). Others have combined participatory approaches with quantitative scenario modelling in the analysis of landscape futures (Walz et al. 2007). Quantitative modelling techniques have included spatial multi-criteria analysis (Pettit and Pullar 2004, Berger 2006), agent-based modelling (Happe et al. 2006), actor-based modelling (Bolte et al. 2004), and integrated assessment and modelling (Carmichael et al. 2004, van Ittersum et al. 2008, Wei et al. 2009). Some studies (Liu et al. 2007, Meyer and Grabaum 2008) have found optimisation and scenario analysis to be a valuable combination for selecting land use and management alternatives under uncertainty.

An application of ***scenario analysis*** global-scale agenda-setting can be seen in the 'Millennium Ecosystem Assessment' (MEA, 2005), which explored the impact of changing ecosystems on human well-being. Three of four scenarios examined in the MEA suggest that "significant changes in policies, institutions, and practices could mitigate some but not all of the negative consequences of growing pressures on ecosystems, but the changes required are substantial and are not currently under way" (MEA, 2005). Given the breadth, magnitude and coarseness of changes suggested in the report, the assessment was not easily translated into finer scale policy and on-ground decision-making. While the MEA findings were successfully integrated into the global programs of the Convention on Biological Diversity, the Ramsar Convention on Wetlands, and the Convention to Combat Desertification, recommendations were less relevant and effective at a local scale, which is where the most tangible changes occur (Tallis and Kareiva 2006). The value of scenarios in that particular instance appears to be more at the agenda-setting phase of the policy cycle, than the policy implementation phase. However, this is not to imply that scenarios, particularly intervention scenarios, cannot be useful in policy implementation.

Developing plausible scenarios helps us take the long view in a world of great uncertainty (Schwartz 1991, Huntley et al. 2010). Scenarios are narratives of the future defined around a set of unpredictable drivers, intended to expand insight by identifying unexpected but important possible directions and outcomes. Scenarios have a timeline over which meaningful change is possible. Scenarios help to develop the means to work towards preferred futures (Huntley et al. 2010, van der Heijden 2005). They are used in long-range planning and the development of robust plans and encompass a broad span of future possibilities so the future can be met with some degree of confidence.

No scenario is ever seen as absolute, as the probability of any scenario being realized is inconceivably small (Stone and Redmer 2006). A challenge of scenario planning is to determine the real needs of corporate leaders and managers. Often, they may not know what they need to know, or may not know how to describe the information that they really want. A value of scenario planning is that

leaders can make mistakes and learn from mistakes without risking important and costly failures in real life (Stone and Redmer 2006). They can make these mistakes in a pleasant, unthreatening, game-like environment, while responding to a wide variety of concretely presented scenarios based on facts. Scenario development often happens in intervention-style processes that may not be tailored to diverse stakeholder demands, or compatible with the need to develop longer-term adaptation pathways that deal with evolving multidimensional challenges because they focus too much on single actions (Berkes and Folke 2002, Wilkinson and Eidinow 2008, Vervoot et al. 2014).

Box 2.3. Case study – Scenario Planning in the Hudson River Estuary Watershed

How a scenario planning exercise typically unfolds in practice is best explored with an example. One such example is The Nature Conservancy's (TNC) 2008 effort to help communities within the Hudson River Estuary Watershed, USA, prepare for the impacts of climate change (see also Cook et al. 2014b for further analysis). In a series of workshops over the course of 18 months, more than 160 stakeholders were consulted, including railroad executives, utility companies, the insurance industry, emergency and health groups, planners and conservation leaders. They identified and discussed important drivers (e.g., land use trends, the political climate) and key uncertainties around those drivers (e.g., will there be strong "top-down" political support for climate change adaptation?). By manipulating these uncertainties and trends, they created four plausible scenarios. Scenarios were described using suggestive titles (e.g., Stagflation Rules) and narrative details, e.g. "the early years of the scenario witness low to negative economic growth, falling real estate values and little new development in the region..." (p. 6). Different elements of each scenario were specified, for example, the projections for the price of gas under the Procrastination Blues scenario were "decline from $3.80 to $2.05 from 2008-2011, then rise rapidly back to $5.00/gal by 2016...". The feasibility of different policies or response options (e.g. changing the requirements for new storm water permits) could then be evaluated, in terms of both the likelihood that they would be adopted in each scenario and how they would perform in each scenario. The 'top performing' options were those that scored relatively highly across the four scenarios. This project provides a good example of the potential of scenario planning for evaluating intervention options. Focussing on the Hudson River Estuary Watershed provided clear geographic scope, the drivers explored were well defined and easily monitored (e.g. the price of gas), meaning that trends within different scenarios could be explicitly and realistically quantified. The response options evaluated were specific enough to be implemented on the ground, for example, developing emergency actions plans with community involvement.

2.3.1.4. Integrative approaches

This last family includes a much larger number of frameworks, approaches and methods than can possibly be described here. For every approach we describe multiple variants exist, often with similar structure and underpinnings, but different names arising from application in different sectors (e.g. forestry, fisheries, transport) or regions. The aim of this section is to provide a brief overview of the sorts of approaches we consider to be integrative. A longer, but far from exhaustive list of approaches can be found in online Appendix #.

Like Multi-criteria decision analysis and AH approaches reviewed above, ***Structured Decision Making*** (SDM; Gregory et al. 2012) derives from multi-attribute utility theory (MAUT; Raiffa 1968). However, SDM also draws heavily on more recent developments in *Decision Analysis* (Raiffa 1968, Keeney 1982, Hammond et al. 1999) and psychology (Kahneman and Tversky 2000). SDM is an organized approach to identifying and evaluating creative options and making choices in complex decision situations. Gregory et al. (2012) define SDM as "the collaborative and facilitated application of multiple objective decision making and group deliberation methods". SDM is designed to deliver insight to decision makers about how well their objectives may be satisfied by potential alternative courses of action. It helps find acceptable solutions across groups, and clarifies divergent values that may underpin irreducible trade-offs. SDM is a very general approach to decision support (Figure 3), which can conceivably be applied to any environmental decision problem at any scale and any level of social and institutional complexity. However, it is the value of SDM in situations in which there are conflicting values and conflicting views about the consequences of various courses of action due to uncertainty that differentiate it from the simpler analytical (or 'normative') approaches (Gregory et al. 2012). The attributes of SDM that distinguish it from the more technical approaches such as MCDA is the emphasis placed on the understanding and dealing with difficult group dynamics through a collaborative approach to clarifying objectives, exploring cause and effect relationships and dealing with contentious trade-offs. To some extent, the application of SDM formalizes or prescribes an approach to dealing with the 'human' elements, including judgement biases, group dynamics, and risk preferences, in decision-making. Tools such as MCDA may be used in an SDM process where they add value or clarity to the process, but the SDM process is not centred on the use of any such tool. See example of SDM in Box 4.

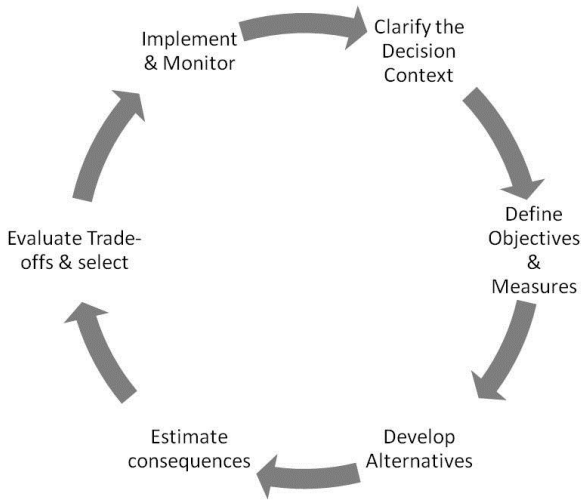

Figure 2.3.: Six basic steps in Structured Decision Making (Adapted from Gregory et al. 2012).

There are six basic steps identified in the SDM framework (Figure 3, Gregory et al. 2012). Clarifying or scoping the decision context involves identifying what the decision is about, which decision(s) will be made, by whom, and when. The spatial and temporal scale over which the decision applies is a key component of clarifying the decision context. Defining objectives and performance measures is a big focus of the SDM approach, which defines what matters in the decision context and how these things will be measured. Objectives and performance measures drive the search for management and policy

options and provide the basis on which they will be compared. The use of objective hierarchies appears to be characteristic of SDM, possibly due to the strong focus on collaboration and encouraging participants to explore, and hopefully better understand each other's values. Developing decision alternatives is a creative, deliberative process that aims to tailor candidate actions (or actions sets) in a way that serves the defined objectives. It is quite common that certain actions most suit the objectives of a particular stakeholder. Evaluating the performance of a particular stakeholder's preferred actions against the criteria of other stakeholders is a key part of understanding the consequences of each alternative. A basic tool used widely in SDM is the consequence table (2.3.2.1) that sets out the expected outcome of each action for each performance measure relating to an objective. The process of estimating consequences of actions for objectives is a key place in which models and scenarios can play a role in SDM. Models and scenarios can help in the exploration of expected outcomes arising from courses of action and the uncertainty about those expected outcomes. Evaluating trade-offs and selecting favoured options then proceeds by considering which options provide reasonable outcomes across all of the objectives considered. Proponents of SDM are generally eager to point out that the evaluation of trade-offs involves "value-based judgements about which reasonable people may disagree" (Gregory et al. 2012). Finally, implementation and monitoring of outcomes enables some post-hoc evaluation of outcomes for the purposes of reporting and learning (McDonald-Madden et al. 2010b); providing opportunity for the SDM process to be adaptive (*sensu* Walters 1986).

Two key strengths of SDM emerge from many of the reported applications. These include the clear separation of facts from values that is at the heart of the approach (Maguire 2004), and the way in which SDM helps with partition and therefore simplify the technical and social complexity that commonly hinders most real-world decision problems. One of the developers of SDM theory and practice describes it as "... the formal use of common sense for decision problems that are too complex for informal use of common sense" (Keeney 1982). This quote highlights the point that there is nothing mysterious or even particularly new about any aspect of SDM, other than the way in which it brings together may key concepts from decision theory toward a workable protocol for deliberations.

A weakness of SDM is that guidance on how to undertake any given step within the SDM 'cycle' tends to be minimal and vague. The key text on SDM for environmental applications (Gregory et al. 2012) emphasizes that the use of SDM is something of an art. Knowing which specific tools to employ in any given decision context at each stage of the SDM process requires significant experience. This means that SDM cannot simply be used 'off-the-shelf' by relatively inexperienced decision analysts. This may be just as well given that poor decisions can have large consequences.

Box 2.4. SDM for non-native fish management in the Glen Canyon dam, USA

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

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

Management strategy evaluation (MSE, sometimes also termed as *Management Procedure Approach*, *Harvest Strategy Evaluation* or *Operating Management Procedures*) uses simulation models within an adaptive framework (*sensu* Walters 1986) to evaluate management options. The objective of the approach is to assess the consequences of alternative management strategies in a virtual world, taking multiple and often competing objectives into account (Butterworth 2007, Bunnefeld et al. 2011). Thus, MSE can be used to reveal the trade-offs in performance across a range of management objectives (Holland 2010). MSE does not prescribes an optimal strategy, instead, it provides the decision maker with the information on which (given their own objectives, preferences, and attitudes to risk) a rational decision should be based. See example of MSE in Box 5.

The conceptual framework and the subsystems modelled by MSE are shown in Figure 4; the modelling steps are discussed based on Rademeyer et al. 2007. An 'operating model' (or preferably, a set of

candidate models) is created to address all of the key biological processes, trade-offs and uncertainties to which an ideal management procedure would be robust (usually one model is chosen as a reference model). These operating models (most typically population dynamics models) are used to compute how the resource responds to alternative scenarios (different future levels of catch or effort). Then performance of each model is integrated over all the considered scenarios. Likelihood of the occurrence of each scenario is regarded as a relative weight given to the output statistics. The final management strategy (procedure) ideally is chosen based on clear, *a priori* defined objectives.

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

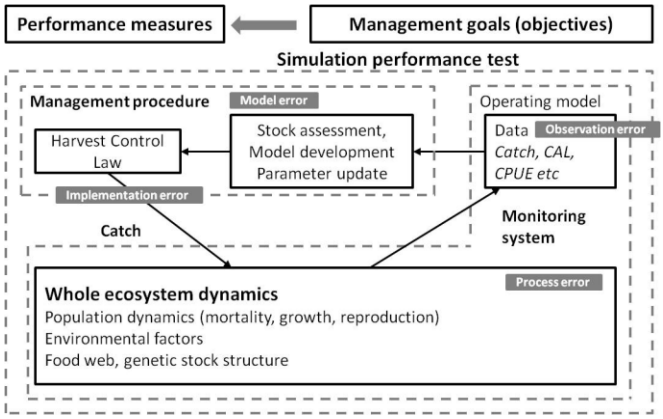

Figure 2.4. The MSE framework (Adapted from Adam et al. 2013, p. 5.)

A core strength of MSE is its transparency and explicit consideration of natural variation and uncertainty in stock assessments and implementation of management controls (Punt and Donovan 2007, Holland 2010). Multiple candidate models are generally considered within simulations to evaluate and test sensitivity to competing hypotheses (Rademeyer et al. 2007). MSE promotes consultation (Bunnefeld et al. 2011) whereby managers and other stakeholders can provide input into the candidate models and scenarios (Nuno et al. 2014). Recent applications have included indigenous interests in the management of socio-economic systems (Plagányi et al. 2013). Technical demands due to complexity and reliance on computer simulation are likely to be the reasons why MSE has not seen wider adoption in fisheries management (de Moor et al. 2011).

Box 2.5. MSE case study – Joint management of fisheries in South Africa

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

Strategic Environmental Assessment (SEA) is the systematic environmental assessment of policies, plans and programmes (Thérivel and Paridario 2013). It can be considered an evidence-based instrument that adds scientific rigour to policy making via suitable assessment methods and techniques (Fisher 2007). The primary aims and objectives of conducting SEA (UNEP 2000) include; (i) supporting informed and integrated decision making by identifying the environmental effects of proposed actions, (ii) considering alternatives and specifying appropriate mitigation measures; and (iii) contributing to environmentally sustainable development, by providing early warnings of cumulative effects and risks (Du et al. 2012).

Strategic Environmental Assessment is now becoming a more frequently used tool with regulations and guidelines for SEA being proposed in many countries worldwide. For example, in the EU the SEA Directive requires an environmental assessment for plans and programmes at national, regional and local levels of jurisdiction. Increasingly, developing countries are introducing legislation or regulations to undertake SEA – sometimes in EIA (Environmental Impact Assessment) laws (e.g. China, Belize, Ethiopia) and sometimes in natural resource or sectors laws and regulations (e.g. South Africa, Dominican Republic). In Australia 'strategic assessments' aim to analyse the cumulative impacts of multiple stressors on species listed as threatened under the Environment Protection and Biodiversity Conservation Act 1999 (EPBC Act). The Convention on Biological Diversity (Article 6b and Article 14) encourages the use of SEA in its implementation (without making it a specific requirement). The Paris Declaration calls for the development of common approaches to environmental assessment generally, and to SEA specifically (www.oecd.org/dac).

EIA legislation often requires an Strategic Environmental Assessment-type approach. However, SEA and EIA are used at different levels of the decision-making hierarchy. The former addresses policies,

plans and programmes; the latter focuses on the lowest hierarchical level (projects). Thus, SEA applies to different geographic and time scales and different levels of detail at strategic and project tiers and it is often conducted before a corresponding EIA. At lower tiers of decision making, SEA is likely to be based on a more rigorous EIA-based approach (such as field surveys, overlay mapping, multi-criteria analysis, and risk assessment), at higher tiers tends to be more flexible, potentially involving methods such as forecasting, backcasting and visioning (Zhang et al. 2004, Liu et al. 2006, Wang et al. 2007, Du et al. 2012). Generally speaking, quantification is more difficult at higher tiers that come with a greater degree of uncertainty (for positive examples see Fischer 2002).

The primary strengths of Strategic Environmental Assessment include the potential for integration of environment and development objectives, reduce the administrative burden of many small-scale impact assessments, reduce the 'death-by-a-thousand cuts' effect of many small impacts by considering cumulative impacts at a regional scale (Hawke 2009), provision for the role of science-based evidence to support informed decisions, capacity to identify and generate new options, potential to build public engagement and improved transparency, increased chance of early problem identification, relative ease of trans-boundary co-operation, and clarity around institutional responsibilities.

On the downside, Strategic Environmental Assessment generally covers a large area and a large number of alternatives, which can make collection and analysis of data complex, operating as the broad scale over a greater range of impacts and values may make SEA less 'certain' than any individual EIA. SEA can be perceived as a risky strategy as potentially large impacts can be approved in a single assessment (Box 6). There are no hard rules about the nature of public consultation under SEA, which opens the method up to minimal or 'token' consultation. Lack of expertise and specialist skills among the general public can lead to power differentials in the process where some stakeholders are well resources, informed and organized. In most administrations under severe human resource and financial constraints, SEA may be seen as a large administrative burden and impossible to properly manage, audit and enforce.

Box 2.6. Strategic Environmental Assessment of the Perth-Peel housing corridor plan in Western Australia.

In 2011, the Australian Federal and Western Australian State Governments formally agreed to undertake a comprehensive strategic assessment of the Perth and Peel Regions of Western Australia in accordance with strategic approvals provisions under the EPBC Act. The strategic assessment was designed to enable a 'big-picture' approach to environment and heritage protection that provides certainty in the long term by determining; (i) areas to be protected from development, (ii) areas suitable for development, (iii) the type of development that will be allowed, and (iv) the conditions under which development may proceed. The primary types of development being considered under the SA were residential, industrial and infrastructure development (roads and rail), as well as expansion of mining and forestry activities over a 30 year planning horizon in a region covering 8,200 km^2 (www.environment.gov.au/node/18607). The assessment process commenced with the generation of maps outlining development options. These options were assessed for their likely impacts on known occurrences and predicted suitable habitat of species listed as threatened under the EPBC Act and relevant state biodiversity legislation. State Government planning and environmental agencies were responsible for developing the strategic plan taking into consideration projected human population growth in the region. A three step biodiversity assessment approach was employed (Figure 5) to help agencies assess the potential impacts of development plans on biodiversity across the region. An iterative process was employed to analyse impacts and refine options, ultimately leading to identification of a lowest impact biodiversity scenario that satisfied development objectives. High value biodiversity areas were characterised using the spatial prioritization software 'Zonation' (Moilanen et al. 2005), based on correlative species distribution models for 61 threatened species and point data for a further 135 species for which there were insufficient records to build distribution models. The strengths of this case study included the development of multiple options which were assessed in an iterative manner, the integration of state-of-the-art biodiversity modelling with a broad scale decision process, and a relatively generous amount of time dedicated to assessing biodiversity impacts of development options prior to decision making.

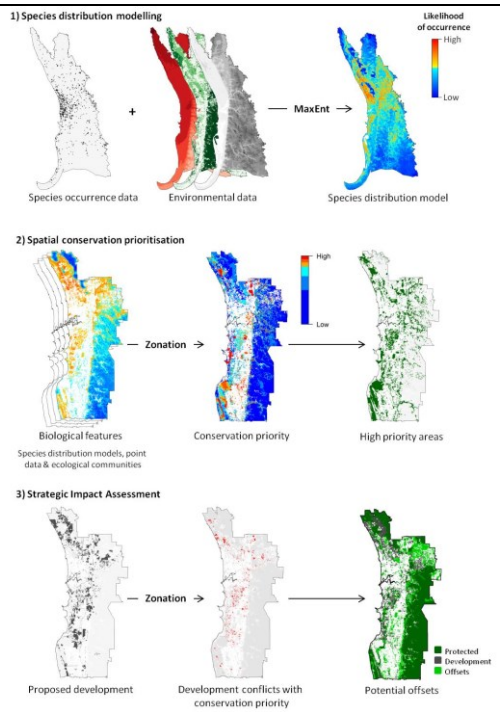

Figure 2.5. SEA analysis framework for assessing the biodiversity impacts of planning options in the Perth-Peel planning region of south-western Australia.

Integrated territorial planning (ITP) is a general and flexible approach to facilitate cooperative planning between neighboring and sometimes overlapping jurisdictions and vertically from the individual land use plot to the national and supranational levels. The aim of territorial planning is to promote common interests or reconcile objectives that could be negatively impacted. ITP seeks to respond to jurisdictions that are recognized by specific national legislation and that are hierarchically organized, such as national, subnational, protected area, private and collective communal land (Amler et al. 1999). Applications of ITP often include establishment of multi stakeholder platforms to facilitate spatial planning across areas that do not respond specifically to jurisdictions, such as watersheds, individual ecosystems or areas of influence of development projects. In this context the strong links across the scales need to be considered in the analysis of land or marine area management (Lambin et al. 2005, Ballinger et al. 2010). Integrated Coastal Zone Management (ICZM) and Integrated Watershed Management (IWM) are examples of territorial planning in specific contexts that are implemented through cross-jurisdictional agreements between representative state, grass roots or private stakeholders (Alves et al. 2011, Ballinger et al. 2010). As ITP tends to be GIS-based, its key strengths lay in its visual products, including thematic maps that can be used across cultures and technical capacities to bridge knowledge systems by presenting both technical information and local knowledge and values.

The ***Delphi technique*** was developed by the RAND Corporation in the late 1960's as a forecasting methodology (Gordon & Helmer 1964, Linstone & Turoff 1975). Soon after, it was adapted as a decision tool (Rauch 1979). Rauch (1979) defines three relevant types of Delphi; Classical Delphi, Policy Delphi and ***Decision Delphi***. The focus of classical Delphi is on forecasting and elicitation; describing the future, while the latter two are focussed on mediating outcomes that influence the

future. Classical Delphi may play a role in agenda setting (*sensu* Chapter 1 of this report), while Policy and Decision Delphi are particularly appropriate when decision-making is required in a political or emotional environment, or when decisions affect strong factions with opposing preferences. Decision Delphi can be used formally or informally to exploit the benefits of group decision making while attempting to insulate against its limitations (e.g deference to authority and groupthink). Example applications of Delphi as a decision tool include allocation of national level health funding in the USA (Hall et al. 1992) and setting priorities for the IT industry in Taiwan (Madu et al. 1991). Delphi can work as an informal, subjective decision support model when the decisions are based on opinion, and can be converted to a formal model, when quantitative data are available.

2.3.3. Matching methods and approaches to the needs of a decision context

In the previous section we explored a sample of decision support approaches and methods under four broad families that to some extent pre-empt the decision context to which they are best suited. A decision context can be conceptualized as a multi-dimensional space defined by axes that are the attributes of decisions listed in Table 2.3. Figure 2.6. provides a graphic conceptualization of a simplified decision context defined using just three such axes; spatial scale, complexity in terms of the number of objectives to be reconciled, and the temporal sequencing of the decision.

Considering the approaches and case studies described above leads to some generalizations about the sorts of decision approaches that lend themselves to application in particular decision making contexts. While some aspects of this relationship between decision context and methods are self-evident; for example, the use of MCDA in decisions involving multiple stakeholders or decision makers, other patterns emerge which may be less obvious *a priori*. For example, it appears that for the most part, sequential decision problems tend primarily to address single-objective problems, while large-scale, multi-objective problems tend not to be handled as sequential, dynamic, adaptive management problems. This may be simply that regional-scale, multi-stakeholder decisions problems tend to be once-off decisions with no plan or program for future changes, or because the inherent complexity of such decisions precludes analysing them as sequential decision problems, even if they are so in reality.

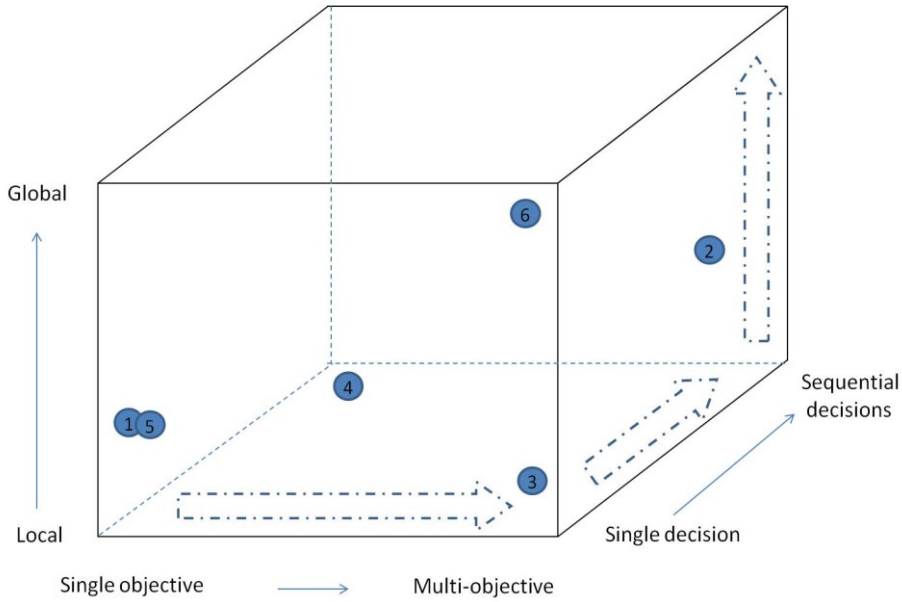

Figure 2.6.: Three dimensions of decision context, with the dashed arrows indicating increasing complexity from a single (once-off) decision made by a single group with a single objective at a local scale, to a sequential decision made by a group of decision makers with multiple (usually competing) objectives at regional/global scales. Numbered circles indicate individual applications of a given decision support method, undertaken in different parts of the decision space. For example, circle 1 represents a study (Joseph et al. 2009) in which a single organization (NZ DoC) used a single objective criteria (maximize increase in species persistence/$) at the national scale regarding. Circle 5 identifies a conservation planning exercise, undertaken by the Malagasy governments, with the single objective of identifying the areas of Madagascar that would most efficiently increase the representativeness of the Madagascan reserve system (Kremen et al. 2008). There was no explicit consideration of sequentially increasing the reserve system or the multiple competing social or cultural objectives in the structured part of the reserve design process, though these considerations would likely have played out in the less structured political process. In contrast, study 2 reports on a decision process in which multiple cultural groups with multiple (incommensurable) objectives participated in a decision about the control of non-native fish species in the Glen Canyon dam in southern USA (Runge et al. 2011a). Study 2 was described as a 'structured decision making' exercise (*sensu* Gregory et al. 2012), supported by MCDA with swing-weighting to help identify dominated options.

The conceptualization of decision context can be extended to the multiple dimensions we believe characterise decision context, though not in a convenient 3-D graphic. We undertook a subjective classification of the sample of decision support approaches we describe above in 2.3.2 above (see Table 2.3. for a stylized representation of the classification and Appendix # for our full analysis) .

Table 2.3.: Decision support approaches and methods assessed against the decision context attributes defined in Section 2.2.2. Attributes are measured in the units described in table 2. The extent to which a particular decision support method or approach is relevant to a particular level of a context variable was subjectively assessed by the authorship group and invited experts. As such, they should be considered at best indicative of the context in which methods *have been* applied and not where they *could* be applied. The table presented here is a summary of the method-by-decision attribute spreadsheet found in Appendix #. The spreadsheet in Appendix X includes example applications of each method, evaluated against the decision context attributes for example applications. Codes used in the table from left to right are: political scale (L=local, N=national, R=regional, G=global), policy cycle (A=agenda setting, P=policy development, I=implementation), type of uncertainty (S=stochastic, E=epistemic, D=decision), decision making entity (public = Pu, private=Pr), decision process (P=participatory, T=top-down), levels of governance (S=single, M=multiple), sectors (S=Single, M=multiple), temporal scope (St=static, Se=sequential), stakeholders (S=single, M=multiple), objectives (S=single, M=multiple), cultural context (h=homogenous, M=multiple), ecological complexity (E=ecosystem, S=species, G=genes), landscape complexity (S=single ecosystem, M=multiple ecosystems), temporal extent (S=short-term, L=long-term), temporal resolution (H=hours, D=decades), spatial extent (L=local, R=regional, G=global), spatial resolution (M=metres, K=kilometres). An asterisk indicates that the method is relevant at any level of the decision context attribute (e.g. can be applied at all spatial scales). An 'x' indicates that the method ignores the specific attribute (e.g. consequence tables do not consider uncertainty).

					Governance attributes								Biophysical attributes					
Method Family	Method	Political scale	Policy cycle	Type of uncertainty	Decision maker	Decision process	Level of governance	Sectors	Temporal scope	Stakeholders	Objectives	Cultural context	Ecological complexity	Landscape complexity	Temporal extent	Temporal resolution	Spatial extent	Spatial resolution
Methods tailored to multi-objective problems	*Consequence tables*	L-R	D,I	x	*	*	*	*	St	x	s	h	*	*	*	*	*	*
	Benefit-cost analysis	L-R	D,I	x	*	*	*	*	St	x	s	h	*	*	*	*	*	*
	MCDA	L-G	P,I	D	*	P	*	M	St	M	M	*	*	*	*	*	*	*
	AHP	L-G	P,I	D	*	P	*	M	St	M	M	*	*	*	*	*	*	*
	Outranking	L-G	P,I	D	*	P	*	M	St	M	M	*	*	*	*	*	*	*
Optimization approaches	*Linear Programming*	L-R	D, I	x	*	T	S	S	St	S	S	h	*	S	*	*	*	*
	Stochastic Dynamic Programming	L-R	D, I	S,E	*	T	S	S	Se	S	S	h	*	S	*	*	*	*
	Goal programming	L-R	D, I	S,E	*	T	S	S	Se	S	S	h	*	S	*	*	*	*
Scenario-based approaches	*Scenario analysis*	*	A	*	*	P	*	M	St	M	M	*	*	*	*	D	*	K
	Scenario planning	L-R	A,P,I	*	*	P	*	M	St	M	M	*	*	*	*	D	*	K
Integrative approaches (often subsume other approaches)	*Structured decision making*	L-R	P,I	*	*	*	*	*	*	*	*	*	*	*	*	D	*	K
	Adaptive Management	L-R	P,I	*	*	*	*	*	*	*	*	*	*	*	*	D	L-R	K
	Management strategy evaluation	L-R	P,I	S,E	*	*	S	S	*	S	S	h	*	*	*	D	L-R	K
	Strategic Environmental Assessment	L-R	P,I	D	*	P	M	S	*	M	M	*	*	*	*	D	L-R	K
	Integrated Territorial Planning	L-R	P,I	D	*	P	M	S	*	M	M	*	*	*	*	D	L-R	K

A key observation across the hundreds of documents reviewed here is that most documented applications of decision support using the methods reviewed above occur at national, sub-national and finer scales. We note that at regional and global scales, there is a rapidly growing number of applications of modelling and scenario analysis in policy agenda setting (e.g. Pereira et al. 2010, Alkemade et al. 2009, Leadley et al. 2010, Leadley et al. 2014, SCBD GBO4 2014), but this has yet to translate into a commensurate rise in the application of structured decision support approaches for biodiversity and ecosystem service policy implementation at those scales. The potential for application of structured approaches to policy development and implementation at the broad scales seems great, but there are clearly strong political, cultural and practical impediments that must be overcome.

2.3.4. Dealing with decision uncertainty

Adequate attention should be given to the role and implications of uncertainty to avoid erroneous policy decisions and adverse consequences. Uncertainties not only in using models and scenarios in decision-making, but also in the process of making decisions itself. The latter can be related to the influence of political "clout" and perceived importance of stakeholder(s), knowledge, values and attitudes of stakeholders, strength of argument presented by stakeholders, values and attitudes of decision-makers, and current political "climate" (Maier and Ascough 2006). The development of uncertainty analysis methods that are able to cater for subjective and non-quantitative factors (e.g. van der Sluijs et al. 2005) is thus critical in making policy decisions on biodiversity and ecosystem services. Both technical and deliberative, participatory methods can be fruitfully applied for dealing with decision uncertainty.

A multitude of technical, mostly mathematical, methods exist for dealing with uncertainty in choice problems (e.g. Stochastic Dynamic Programming, robust optimization, Info-gap decision theory, etc.). An advantage of technical decision support approaches to dealing with uncertainty is that the role of modelling is clearly defined. Without a system model describing the dynamics of the system, uncertainty analysis is impossible. Use of uncertainty analysis methods demand that the analyst is explicit about the uncertainties impacting a particular decision. Even if exact optimal solutions do not meet manager and stakeholder expectations since all of their concerns can seldom be incorporated, application of these formal uncertainty analyses does highlight which uncertainties are most important and which are inconsequential. This can provide a strong motivation and guidance for investing in reduction of critical uncertainties. The primary impediment to the use of these models is the high technical expertise needed and limitations on the number of state variables (and states) that can be handled in practice.

Decisions on biodiversity and ecosystem services are often characterised by "wicked" problems, where ecological complexity is compounded by social complexity, involving multiple, active, stakeholder groups with diverse values operating in uncertain and shifting administrative, economic, political, and legal environments (Balint et al. 2011). Such problems require deliberation among decision makers and stakeholders to allow learning throughout the decision-making process. There are only a few participatory approaches that allow deliberation to deal with the complexity of both nature- (stochastic) and human-related (decision-making) uncertainties in ecosystem management.

The most frequently and widely applied approaches include, for example, Scenario Planning Approach (SPA), Collaborative Adaptive Management Approach (CAMA), and Deliberative Multi-Criteria Evaluation (DMCE).

Both technical and deliberative approaches to dealing with uncertainty in decision making often refer to, and draw on the concept of ***adaptive management*** (*sensu* Walters and Holling 1983). Box 2.7. below provides an example of a single-jurisdiction, dual objective decision analysis problem, involving stochastic and epistemic uncertainty that are addressed using a relatively technical strategy for learning and decision making based on stochastic dynamic programming (2.3.1.2). This relatively technical approach to adaptive management has been utilized in fisheries (Hilbourne 1992), threatened species management (McDonald-Madden, Rout), and invasive species management (Shea et al. 2002). Decision makers and policy analysts commonly invoke the adaptive management concept as a valuable heuristic supporting continuous improvement, though it is not always clear whether such applications explicitly include a plan for learning, which some see a fundamental aspect of adaptive management. See example of how to deal with uncertainty in Box 2.7.

Box 2.7. Dealing with uncertainty using adaptive management - Management of North American Mallard ducks

Nichols and Williams (2006) summarise an adaptive management program that has been working for 10 years to support the management (hunting regulations) of mid-continent Mallard ducks in North America. Management objectives are to maximize cumulative harvest over a long time period (including the devaluation of harvest when predicted population size falls below the North American Waterfowl Management Plan goal threshold of 8.8 million breeding mallards). Management actions include four regulatory packages specifying daily bag limits and season lengths for each of the four major North American flyways. Four models (scenarios) of system response to harvest management are included in the model set. These models reflect two different hypotheses about the effect of hunting mortality on annual duck survival (compensatory mortality reflecting minimal effects of hunting and additive mortality reflecting maximal effects of hunting mortality), and two hypotheses about the strength of density-dependent relationships defining reproductive rates (weakly and strongly density-dependent). At the initiation of this management process in 1995, all four models (representing all possible combinations of these four hypotheses) were given equal credibility weights of 0.25, indicating no greater faith in the predictions of one model than in those of any other. A complex monitoring program is used to estimate breeding population size and number of wetlands in Prairie Canada (an important environmental covariate), rates of survival and harvest, and preseason age ratio. In each spring, the new estimate of population size is compared against predictions made the previous spring corresponding to each of the four models. These comparisons are combined with the model weights from the previous year to update the weights. Learning thus occurs when weights become large for some models, giving them more credibility and thus more influence in the decision process, and small for others. The decision about which set of harvest regulations to implement depends on system state, as defined by estimated numbers of ducks and ponds.

2.3.5. Reflection on decision support approaches: ingredients for success and impediments to adoption

Reflecting on the sample of policy, planning and management support decision approaches and case studies we have reviewed, a couple of observations about success emerge. A defining feature of the documented successful applications of decision support appears to be the level of commitment and involvement of decision analysts or facilitators for the duration of the decision process. In the previous sections we have documented successful examples of decision processes that ranged from highly participatory, deliberative, non-technical exercises (e.g. the Tacana land management exercise), to more technical exercises (e.g. the Paijanne Lake MCDA exercise in Finland), and combinations of the two (e.g. non-native fish control in Glen Canyon). All had very strong commitment and support from decision analysts, modellers, and/or facilitators. These people might be considered 'champions' of their given decision support approach or method, and like champions of change, they are essential for successful use of models and scenarios in formal decision processes (Guisan et al. 2013).

Nonetheless, there is a mismatch between the preponderance of academic and theoretical studies around modelling, scenario development, and decision support approaches, and the relatively small number of documented case studies that present successful application of models and scenarios in decision making in the environment sector, especially at the broader regional and global scales. It is hard to imagine that the relatively small number of documented successful examples of models and decision-making is due solely to a lack of champions. By comparison to biodiversity and ecosystem services, examples of successful application of formal decision approaches such as MCDA and scenario planning (often using models and scenarios) in other sectors such as manufacturing, business, and the military abound. There appears to be some particular impediment to wider application of such approaches in biodiversity and ecosystem service policy, planning and management. It may relate to the complexity of such systems, a general lack of trust in data and measurement methods, or a willingness to invest the time and financial resources into making it work well. Accessibility of data and models to decision makers and stakeholders is an issue recognized at all scales. This issue can be partly addressed by data and interface development (addressed in Chapter 7 and 8). Model outputs often fail to meet decision-making objectives. This can be partly addressed by improving communication and expectations about the capacity of models to deliver the information most relevant to decisions, and by improvements in the capacity of models to deliver, though technical and conceptual advances.

A lack of appreciation of the potential role of models on behalf of decision makers is another impediment to uptake that can be partly overcome through improved communication and better documentation of successful application of models, scenarios and decision support. Exploration of methods to improve the credibility of model predictions by collection of empirical evidence demands further attention. The capacity of models to sensibly characterise uncertainty is a key component of their credibility.

The issue of spatial scale appears to be a dominant challenge in relation to the use of models and scenarios and decision support processes. Many encouraging examples of the application of models and scenarios within formal decision processes at local-national scales can be found in published literature. As we move to regional (multi-national) and global, a small but growing number of

examples exist in which models and scenario have been used to set the policy agenda by highlighting key future challenges to biodiversity and ecosystem services. However, this has not led to widespread adoption of decision support approaches at those scales that provide a means to ensure that model and scenarios outputs are used efficiently and properly. Several possible reasons exist for the lack of compelling examples of structured decision approaches at global and regional scales. Political forces may not be completely comfortable with 'handing over' complex and sensitive decisions to technocrats using systems that policy makers don't fully understand or trust, or they may be uncomfortable with the level of transparency about motivations, values and scientific facts that structured approaches bring to decision-making. This implies several key challenges. There is the challenge of educating policy makers to understand that involvement in decision processes doesn't have to mean relinquishing power. Conveying the notion that structured approaches to decision-making that judiciously utilize models and scenarios can help reduce complexity, distil true differences of opinion and values from linguistic ambiguities or confusion, increase mutual understanding of each others' values, and reduce conflict.

2.4. Information needs

The information needs of policy and decision makers, including consideration of which uncertainties are prominent in deliberations, determine the modelling approaches, and intervention or exploratory scenarios most likely to deliver information pertinent to the decision or policy at hand. Here we explore the means by which those information needs are determined and how the information needs inform the choice of models and scenarios.

2.4.1. What do we want from our models?

Model output requirements vary from assessment to assessment and decision to decision. Biodiversity model outputs range from predictions about the spatial and temporal variation in the prevalence of a gene or abundance of individuals of a species (e.g. GENE mapping, Keith et al. 2008) to community compositional dissimilarity to coarse representations or indices of species richness or mean species abundances (e.g. Alkemade et al. 2009) (See Chapter 4 and 5) under one or many scenarios. Ecosystem service model outputs are similarly diverse in the type, complexity, spatial extent and resolution of outputs.

The choice of model output(s) for a given decision or assessment should be determined by a clear articulation of the objectives of the decision or assessment. The importance of articulating clear and measurable objectives has been emphasises in the decision science literature for decades and a number of tools exist to assist in articulating objective, such as objectives hierarchies. The most common problem that arises when aligning model outputs to the fundamental objectives of an assessment of decision is that objectives have not be clearly articulated, or they are embodied in vague statements such as "ensuring a sustainable future for Springfield" for which there may be a huge set of possible model outputs or indicators. Another common impediment to choosing model outputs that are proximal to fundamental objectives is that the measures prescribed in the objective statement are impossible or highly impractical to model. For example, a regional level objective to

secure all remaining mammals in the amazon basin while increasing economic opportunities for local peoples could conceivably be supported by models of the population viability analyses (PVAs) of all mammals and socio-ecological models of local livelihoods in the amazon basin under a range of future climate, land-use and intervention scenarios. However, it is highly unlikely that PVAs for every Amazon Basin mammal could be constructed in time to influence any decision process. For this reason, surrogate model outputs that can be developed within time, budgetary, and expertise constraints are commonly used to approximate the ideal measure of the fundamental objective.

For terrestrial ecosystems at almost all spatial scales, the most commonly used surrogates of biodiversity are species distribution models (SDMs) and various aggregations of SDMs. SDMs are appealing for the reason that observation data and mapped environmental variables are readily and freely available, the technology to fit and evaluate SDMs is readily available and easy to use, and large numbers of species can be processed rapidly. With the many benefits of distribution modelling come many limitations and drawbacks that are well documented (REFS). For the purposes of characterising and predicting long-term biodiversity persistence they are a useful, if blunt instrument. For many reasons described in the ecological modelling literature (Fordham et al. 2011; Dormann 2009; Thuillier et al. 2004), they do not adequately characterise many of the spatial and temporal processes that mediate persistence or extinction in changing environments. For example, failure to explicitly deal with dispersal limitations, competition and predation, or plasticity and evolution of thermal and other niches, they may be missing much that is important in the extinction process. However, by representing spatial and temporal variation in the availability of suitable habitat, they provide a distal surrogate for species persistence over medium to long time-frames. If combined with some other coarse analyses (e.g. Carrol et al. 2010) that bring in some consideration of spatial processes, they may provide some useful information about the relative merits of alterative conservation options under environmental and land-use change. While it is desirable for model outputs to directly reflect the fundamental objectives of a given assessment of decision problem, in many instances it will not be possible, and in such instances surrogate outputs (e.g. SDMs) are often better than nothing. Careful consideration of exactly what value model outputs bring to an assessment or decision problem is a necessary ingredient for successful integration at the decision/modelling interface. Failure to take good care with planning outputs for decision problems is likely to result in a breakdown (or continued disengagement) between modellers and decision makers (Addison et al. 200X).

2.4.2. What scenarios need to be explored?

The *need* for scenarios is generated at the 'assessment and decision support interface' (See Figure 1.3 in Chapter 1) given the assessment/policy/planning/management problem at hand. For example, if a coastal management body needs to make decisions about where to allow housing development, they may need carbon emissions scenarios to underpin modelling of sea-level rise in their region and to characterise uncertainty about future sea-levels relating to emissions. A regional biodiversity assessment is likely to require human population growth and land-use change scenarios to inform biodiversity models used to make projections of biodiversity change over several decades.

Two broad classes of scenario arise; exploratory scenarios and intervention scenarios (aka 'goal-oriented' scenarios) (See Chapter 1 and 3). Exploratory scenarios as used to explore the sensitivity of performance variables (e.g. species persistence or fresh water availability) to a range of plausible or

possible futures. How exploratory scenarios are determined and how many can or should be considered is open to the interests, concerns, and imagination of the participants in any given assessment or decision problem. Scenarios can be generated by asking; "What future contingencies are likely to impact on the environmental assets, goods and services from our region that we value?" or through more formal or structured means of scenario elicitation (Carpenter et al. 2006). Exploratory scenarios have been applied in assessments at all spatial scales from local to global (Alkemade et al. 2009; MEA 2003 and 2005a). Intervention scenarios represent possible or anticipated futures arising under a set of specified interventions. Interventions can take the form of management actions, policy options, planning options, and should be specified within realistic social and economic constraints so that they can be considered plausible futures given a certain policy pathway (e.g. Sandker et al. 2009). While intervention scenarios fit most naturally in the domain of decision analysis, they may still provide interesting and motivating input to assessments and may play a crucial role in agenda setting (*sensu* Chapter 1).

2.4.3. Characterizing and communicating uncertainty to decision makers

A big challenge still remains in conveying the importance of communicating uncertainty, which should go far beyond the traditional statistical approaches (Ramos et al. 2010). Various wasy of communicating and presenting uncertainty have been developed, but not all are easy to understand by non-technical audiences, which may lead to misinterpretation (Wardekker et al. 2008). Existing uncertainty guidance documents range from being very generic and broad to more specific guidance documents (Van der Jeur et al. 2010). Examples of specific guidance documents are the IPCC Consistent Treatment of Uncertainties (Mastrandres et al. 2010), Guide for Uncertainty Communication (GUC) (Peterson et al. 2013), and Progressive Disclosure of Information (PDI) (Wardekker et al. 2008). The IPCC uses words to reflect different levels of probability (e.g. very unlikely for 0-10% probability, virtually certain for 99-100% probability, etc.) and levels of confidence (e.g. very low, very high, etc.). Using words is useful for 'deep' uncertainties (e.g. qualitative issues, such as problem framing, choice of methods, general level of knowledge and value-ladenness) that cannot be easily quantified or expressed probabilistically and are hard to communicate using traditional methods, such as probability terms, uncertainty ranges, and error bars (Wardekker et al. 2008). The GUC suggests three sets of questions relating to i) audience and relevance, ii) distribution of information and iii) presentation of information, which researchers should answer when communicating about uncertainty in written reports and presentations.

The PDI is a systematic approach for distributing uncertainty information to audiences and classifies uncertainties into outer and inner layers depending on which medium of communication is used. The outer layers are non-technical information (e.g. press release, summary) which integrates uncertainties into the message and gives emphasis on context, implications and consequences, while the inner layers are detailed technical information (e.g. appendices, background report) which discusses uncertainties separately and emphasise types, sources and the extent of uncertainty (Petersen et al. 2013, Wardekker et al. 2008). Information disclosure is considered concrete operationalization of transparency in the global environmental domain and applied in various international, inter-governmental, private and NGO "governance-by-disclosure" initiatives (Gupta 2008). A very relevant international example is the Aarhus Convention that aims to enhance access to

information, public participation in decision-making and access to justice in environmental matters (UNECE 1998). In the private sector, public disclosure of corporate environmental information is an informal policy tool to overcome the ineffectiveness of regulative and market-based approaches to protect the environment (Anbumozhi et al. 2011).

2.5. Gaps, knowledge needs and capacity building

A huge number of agenda-setting and decision support approaches now exist that have been used in a wide variety of policy contexts, utilizing a wide variety of biodiversity and ecosystem service models and scenarios. On reviewing a large number of assessment and decision processes, we find that a key ingredient for the successful application of structured decision support, models and scenarios is the dedication and continuity of involvement of decision support facilitators and modellers in close collaboration with decision makers and decision making processes. We find that the primary impediments to the widespread and productive use of models and scenarios in policy, planning and management is a general lack of trust in modellers, models and scenarios, a lack of understanding among decision makers about the positive role that models and scenarios can play, a general lack of decision support, modelling and scenario analysis skills relative to the number of policy, planning and management processes, a lack of data to underpin the models and scenarios of most interest to policy makers and managers, a lack of willingness on the part of modellers to engage fully in real-world decision problem and develop the most relevant scenarios and models for the problem at hand, a lack of transparency in approaches to modelling and scenario development, and complex political agendas that are not amenable to the transparency ideally associated with good modelling and scenario analysis. Increased collaboration between modellers and decision makers will lead to increased trust, better and more relevant models and scenarios, and a culture of decision support based on models and scenarios that is robust to complex political agendas.

There are several elements of capacity that must be built if the gap between modellers, decision analysts, and decision makers is to be bridged. Capacity in modelling and decision analysis appears to be concentrated in certain geographic regions. Provision of training, hard-ware and software to facilitate increased capacity in modelling, scenario analysis, and decision support in developing nations is urgently needed.

However, the challenge of increasing uptake of decision support approaches is not only based on the availability of technical expertise and equipment. A larger challenge is a cultural one. The capacity of modellers and decision analysts to influence decision processes in a positive way is impaired throughout the world by poor communication, and the fact that much of the skill base resides in academic institutions, for which there are little tangible rewards for being involved in real decision processes.

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