Chapter 1

Integrating the ecological and economic dimensions in biodiversity and ecosystem service valuation

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Contents

Key messages

- Linking biophysical aspects of ecosystems with human benefits through the notion of ecosystem services is essential to assess the trade-offs (ecological, socio-cultural, economic and monetary) involved in the loss of ecosystems and biodiversity in a clear and consistent manner.

- Any ecosystem assessment should be spatially and temporally explicit at scales meaningful for policy formation or interventions, inherently acknowledging that both ecological functioning and economic values are context, space and time specific.

- Any ecosystem assessment should first aim to determine the service delivery in biophysical terms, to provide solid ecological underpinning to the economic valuation or measurement with alternative metrics.

- Clearly delineating between functions, services and benefits is important to make ecosystem assessments more accessible to economic valuation, although no consensus has yet been reached on the classification.

- Ecosystem assessments should be set within the context of contrasting scenarios - recognising that both the values of ecosystem services and the costs of actions can be best measured as a function of changes between alternative options.

- In assessing trade-offs between alternative uses of ecosystems, the total bundle of ecosystem services provided by different conversion and management states should be included.

- Any valuation study should be fully aware of the ‘cost’ side of the equation, as focus on benefits only ignores important societal costs like missed opportunities of alternative uses; this also allows for a more extensive range of societal values to be considered.

- Ecosystem assessments should integrate an analysis of risks and uncertainties, acknowledging the limitations of knowledge on the impacts of human actions on ecosystems and their services and on their importance to human well-being.

- In order to improve incentive structures and institutions, the different stakeholders - i.e. the beneficiaries of ecosystem services, those who are providing the services, those involved in or affected by the use, and the actors involved at different levels of decision-making - should be clearly identified, and decision making processes need to be transparent.

1 Introduction

In spite of the growing awareness of the importance of ecosystems and biodiversity to human welfare, loss of biodiversity and degradation of ecosystems still continue on a large scale. Fundamental changes are needed in the way biodiversity, ecosystems and their services are viewed and valued by society. A major difficulty is that many ecosystem[i] services [ii] are (mixed) public goods, and use levels are therefore difficult to regulate, even when they are at or near the point of exhaustion. Although many people benefit from ecosystem services, individuals or groups usually have insufficient incentives to maintain ecosystems for continued provisioning of services. For example, open access fisheries provide valuable harvests but often suffer from over-exploitation that leads to declines in fish populations and lowered future harvests.

The problems of management and governance of ecosystems stem from both poor information and institutional failures. In some cases, knowledge is lacking about the contribution of ecosystem processes and biodiversity to human welfare and how human actions lead to environmental change with impacts on human welfare. In other cases institutions, notably markets, provide the wrong incentives.

These two types of failures, and the complex dynamics between the ecology-economy interface, often lead to large scale and persistent degradation of the natural environment and accelerating loss of ecosystem services and biodiversity. Given the large scale of human activities on the planet, the point has been reached where the cumulative losses in ecosystem services are forcing society to rethink how to incorporate the value of these services into societal decision-making.

The release of the Millennium Ecosystem Assessment (MA 2005a) helped foster use of the concept of ecosystem services by policy makers and the business community. However, progress in its practical application in land use planning and decision making has been slow (e.g., Daily et al. 2009, Naidoo et al. 2008).

This lack of progress stems not only from failures of markets and systems of economic analysis and accounting (notably GDP) to capture values of ecosystem services, but also from our limited understanding of: a) how different services are interlinked with each other and to the various components of ecosystem functioning and the role of biodiversity; b) how different human actions that affect ecosystems change the provision of ecosystem services; c) the potential trade-offs among services; d) the influence of differences in temporal and spatial scales on demand and supply of services; and e) what kind of governance and institutions are best able to ensure biodiversity conservation and the sustainable flow of ecosystem services in the long-term.

Without changes in institutions and incentives, further declines in natural capital are likely, as those who gain from actions that deplete natural capital will continue to avoid paying the full costs of their actions and pass these costs to poor societies and future generations (Srinivasan et al. 2008). Although such estimations are fraught with difficulties, it can be argued that the cumulative loss of '*natural capital*' (see further) over the past decades has cost, and still costs, the global community large sums of money in terms of damage, repair and replacement costs (Bartelmus 2009).

One of the aims of the TEEB study is to provide more and better data and understanding of the (economic) significance of these losses and the consequences of policy inaction on halting biodiversity loss at various scales (global, regional and local). Although emphasis is on the economic, notably monetary, effects of the loss of ecosystem services, TEEB will give due attention to the underlying changes in ecological 'values' (ecosystem integrity and life-support functions) and socio-cultural implications.

This TEEB D0 report provides the science basis of the economics of ecosystems and biodiversity, whereas the other TEEB products (D1-D4) are dedicated to specific audiences, including policy makers, administrators, business or consumers.

Chapter 1 summarizes recent developments and describes our TEEB framework, building upon the TEEB Interim report (European Commission, 2008) to further operationalise the economics of biodiversity and ecosystem services. Each step within the framework roughly coincides with a chapter in the entire D0 report.

Chapters 2 and 3 explore the ecological basis of the assessment. Chapter 2 presents our current state of knowledge on the relationships among biodiversity, ecosystems and ecosystem services. It provides an elaboration of relevant ecological concepts, and highlights the uncertainties and risks associated with ecosystem change caused by humans at an ever-increasing pace. Once biodiversity is lost and ecosystems have irreversibly changed, it can be very expensive or no possibilities may remain to restore these systems and recover associated ecosystem services. Chapter 3 provides a review of existing biophysical measures and indicators that are used to quantify and map current knowledge on biodiversity and ecosystem services, including their merits and shortcomings. Research efforts are needed towards better indicators, especially for measuring changes in biodiversity and the provision of services to serve as a basis for economic valuation.

Chapters 4 to 6 address the 'human well-being' component of the framework. Chapter 4 establishes the basis for a much-needed encompassing understanding of valuation, including ecological, economic and social values, before it more specifically discusses the social and cultural contexts of biodiversity and ecosystem service valuation. Chapter 5 then provides a detailed discussion of the merits, issues and challenges to applying (i) monetary valuation techniques and then (ii) benefits transfer in the context of this assessment. In Chapter 6, some of the ethical issues for economic valuation are explored, in particular the use and selection of discount rates that have to be critically reconsidered both with respect to ecological uncertainties and distributional equity.

Chapters 7 to 9 use the framework to make an analysis of the economic implications of the loss of ecosystem services while taking into account the methodological recommendations developed in chapters 2 to 6. It should be realized up front that data and methodological limitations remain, and the TEEB assessments provided are therefore chosen to illustrate the benefits and costs of biodiversity conservation and sustained ecosystem service provision in different contexts and at different spatial scales. It is not the intention to make an estimate of the total economic value of ecosystem services at a global scale. Chapter 7 presents a synthesis of the empirical economic valuation literature in the form of a matrix of values for the main types of ecosystems and ecosystem services. Based on the analysis of the previous chapters, ranges of values are selected that are relevant for different geographic regions and income levels and collected in a database, that is then used in chapters 8 and 9. Chapter 8 makes a preliminary analysis of the costs of action and inaction for several biomes including terrestrial and marine. This analysis is based on a review of available models and recent global assessments, on a selection of scenarios, and on combining various methodologies and sources for the monetary valuation of benefits and costs. Chapter 9 explores the state of knowledge on assessing the impacts of changes in ecosystem services and comparing the costs of inaction and action at the macroeconomic scale.

Finally, *chapter 10* draws key conclusions and recommendations from the TEEB D0 report.

2 Review of existing frameworks linking ecology and economics

Over the past few decades many attempts have been made to systematically link the functioning of ecosystems with human wellbeing. Central elements in this "link' are the intertwined notions of natural capital 'stocks' and the ecosystem services that flow like interest or dividends from those stocks. According to the Millennium Ecosystem Assessment (MA 2005a), natural capital is "an economic metaphor for the limited stocks of physical and biological resources found on earth". The continuing depletion and degradation of natural capital has generated concerns and debate over the capacity of the economic system to substitute for these losses with human-made capital, and the conditions for sustainable development, defined as non-declining welfare over generations (Pezzey 1992; Pearce et al. 1989). While the degree of substitutability is ultimately an empirical question, it is generally recognised that substitution has limits (Barbier 1994; Daly 1996; Prugh 1999; Daly and Farley 2004), and that a critical amount of natural capital has to be preserved (see also chapters 4, 5 and 6 for the implications of this debate for the TEEB study).

This section provides a brief overview of the development of the theory and practice of ecosystem functions and services and discusses some key insights and challenges from the literature and the TEEB Interim Report (2008).

2.1 Ecosystem services: early developments and recent frameworks

Thinking about people-environment interactions and their effects on human welfare stretches back centuries and includes writings from Roman times on the increase in population and decline in what we now call ecosystem services (Johnson 2000). Early modern writers on the subject include Marsh (1874), Leopold (1949), Carson (1962), and Krutilla and Fisher (1975) to mention but a few. In 1977, Westman published a paper in *Science* examining the link between ecological and economic systems entitled “How much are Nature’s Services Worth?” (Westman 1977). Ehrlich and Ehrlich (1981) later coined the term ‘ecosystem services’ and in the following decade ecologists further elaborated the notion of ecosystems as life-support systems, providers of ecosystem services and economic benefits (see for example Ehrlich and Mooney 1983; Odum 1989; Folke et al. 1991; De Groot 1992). But it was not until the late 1990’s that the concept got widespread attention with the publications by Costanza et al. (1997) and Daily (1997). At the same time, the interdisciplinary field of ecological economics developed the concept of natural capital (Costanza and Daly 1992; Jansson et al. 1994; Dasgupta et al. 1994), which includes non-renewable resources, renewable resources and ecosystem services to demonstrate the significance of ecosystems as providing the biophysical foundation for societal development and all human economies (Common and Perrings 1992; Arrow et al. 1995). In an attempt to facilitate discussion and systematic analysis of ecosystem services, De Groot et al. (2002) created a classification system specifying the relationship between, and transitions from ecosystem processes and components and their transition to goods and services.

Based on these and other studies[iii], the Millennium Ecosystem Assessment (MA 2005a) recognized four categories of services: supporting (e.g. nutrient cycling, soil formation and primary production); provisioning (e.g. food, fresh water, wood and fiber and fuel); regulating (e.g. climate regulation, flood and disease regulation and water purification); and cultural (aesthetic, spiritual, educational and recreational) (see Figure 1).

The introduction of the concept of ecosystem services on the global agenda by the MA provides an important bridge between the imperatives of maintaining biodiversity and the challenges in meeting to meet the Millennium Development Goals.

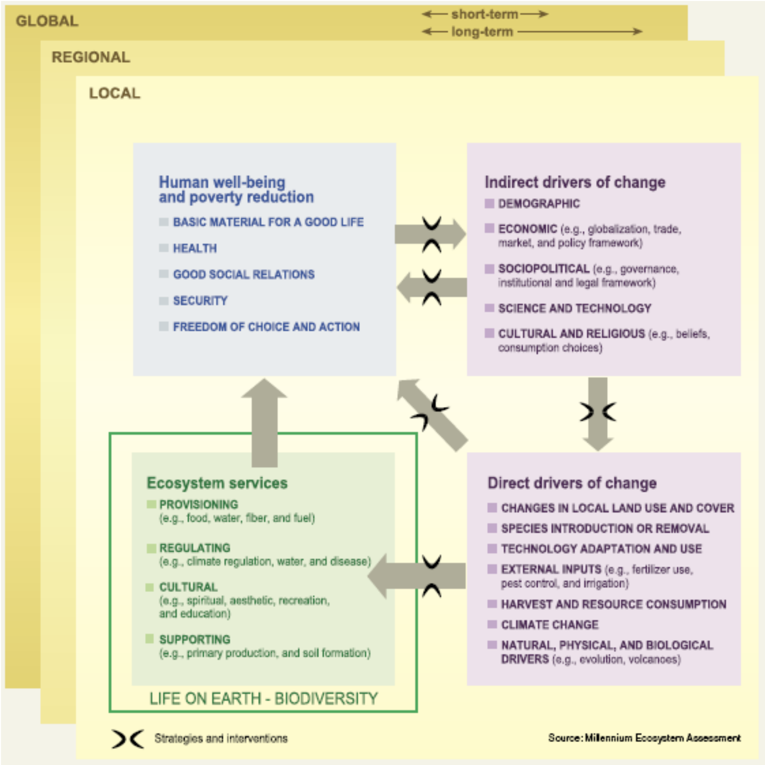

Figure 1: **MA conceptual framework: linking ecosystem services and human well-being**
Source: MA (2005a).

2.2 The TEEB Interim report and further recent frameworks

The Millennium Assessment, purposely, did not pay much attention to the economics of ecosystem change and in the preparation phase of the TEEB study (TEEB 2008), a framework was therefore proposed (see Figure 2) for articulating the ecological and economic aspects of the analysis necessary for the valuation of biodiversity loss and ecosystem degradation (Balmford et al., 2008).

This scheme stresses the need to rely on counterfactual scenarios that differ through specific actions aimed at addressing the main drivers of loss. Changes in the delivery of services need first to be estimated and mapped in biophysical terms, which requires a sufficient understanding of the factors that drive their production and how they are affected by the actions put in place. Economic valuation should then be applied to the changes in services, which requires a good understanding of the service flows and of the determinants of demand.

Being spatially explicit is important in order to take into account the spatial heterogeneity of service flows and of the economic values that can be assigned to them, as well as the variability of conservation costs. It also allows the identification of mismatches of scales as well as analyzing the distributional implications of decisions that affect ecosystems, and exploring trade-offs.

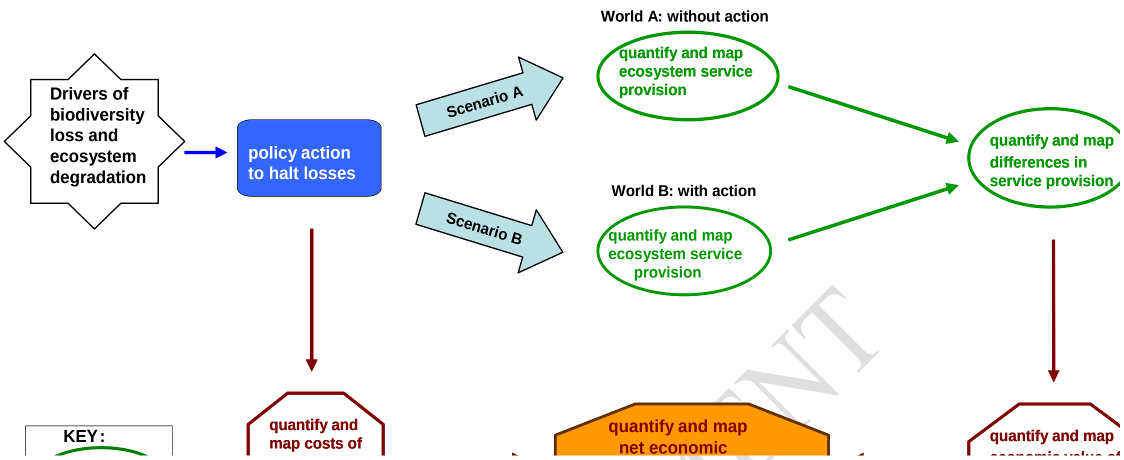

Figure 2: An economic valuation framework: contrasting states of the world
Source: modified after Balmford et al. (2008) and TEEB (2008).

Marginal valuation in economic thinking assumes substitutability between services and is therefore only applicable within certain ecological limitations, requiring that no irreversible ecosystem changes occur (see chapters 2 and 4 for more detail). Next to these ecological limitations, socio-cultural considerations may delimit the range of valid cases for marginal valuation (Turner et al. 2003). Therefore, any valuation of biodiversity and ecosystem services needs to take account of the range of ecological and socio-cultural values that are not covered by economic valuation, but need different approaches and methodologies to be reflected in decision making (EPA-SAB 2009).

The TEEB Interim report's valuation framework is largely consistent with others recently proposed in the analysis undertaken by the US National Research Council (NRC 2005), including the Natural Capital Project (Daily et al. 2009), the EPA Science Advisory Board (EPA-SAB, 2009), Valuing the Arc (Mwakalila et al. 2009), and the French Council for Strategic Analysis (Chevassus-au-Louis et al. 2009). In all of these efforts, the essential links between human actions, ecosystems, services and their contributions to human welfare are (see Figure 3, building on Daily et al. 2009). Human decisions lead to actions that have impacts on ecosystems, causing changes in ecosystem structure and function. These changes in turn lead to changes in the provision of ecosystem services. Changes in ecosystem services have impacts on human welfare. A clear understanding of these links can provide information that can lead to the reform of institutions and better decisions that ultimately improve the state of ecosystems and the services they provide to society.

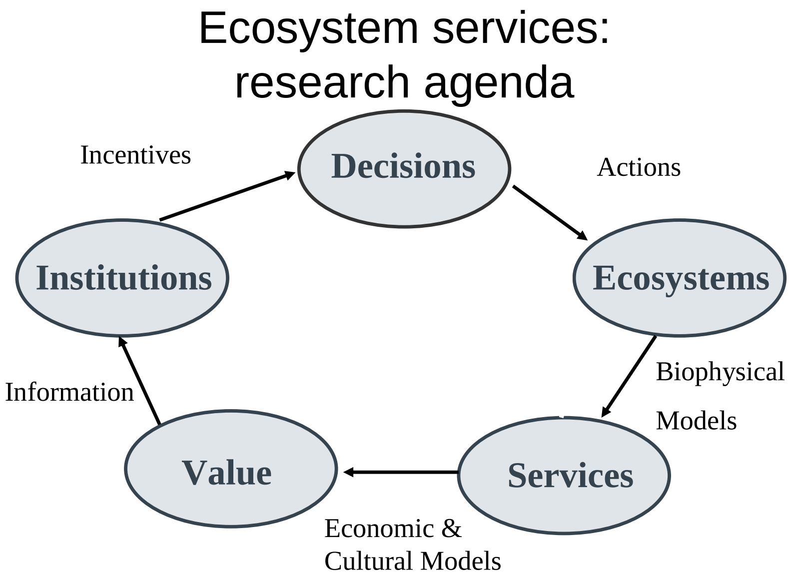

Figure 3: **Ecosystem Services: research agenda**
Source: Daily et al. (2009).

2.3 Defining ecosystem functions, services and benefits

Research efforts regarding the investigation of ecosystem services have increased strongly in the past ten years (Fisher et al. 2009). They have provided much insight in how to ensure that ecosystem service research is scientifically robust and credible, and also conveys a clear message to decision-makers in both the public and private sectors.

In spite of the work done so far, there is still much debate about definitions and classifications (e.g., Daily 1997; Boyd and Banzhaf 2007; Wallace 2008; Costanza 2008; Fisher and Turner 2008; Fisher et al. 2009; Granek et al. 2009) and perhaps we should accept that no final classification can capture the myriad of ways in which ecosystems support human life and contribute to human well-being. Yet for a global assessment like TEEB, it is essential to be clear about the terminology and classifications used. When dealing with complex relationships like coupled social-ecological systems, we need a rich language to describe their different features and interactions. While accepting that no fundamental categories or completely unambiguous definitions exist for such complex systems, and that any systematization is open to debate, it is still important have to be clear about the meaning of the core terms used.

Figure 4 gives a schematic representation of the way TEEB proposes to disentangle the pathway from ecosystems and biodiversity to human wellbeing. A central concept in this diagram is the notion of (ecosystem) service which the MA defined simply as "the benefits humans derive from nature" (MA 2005a).

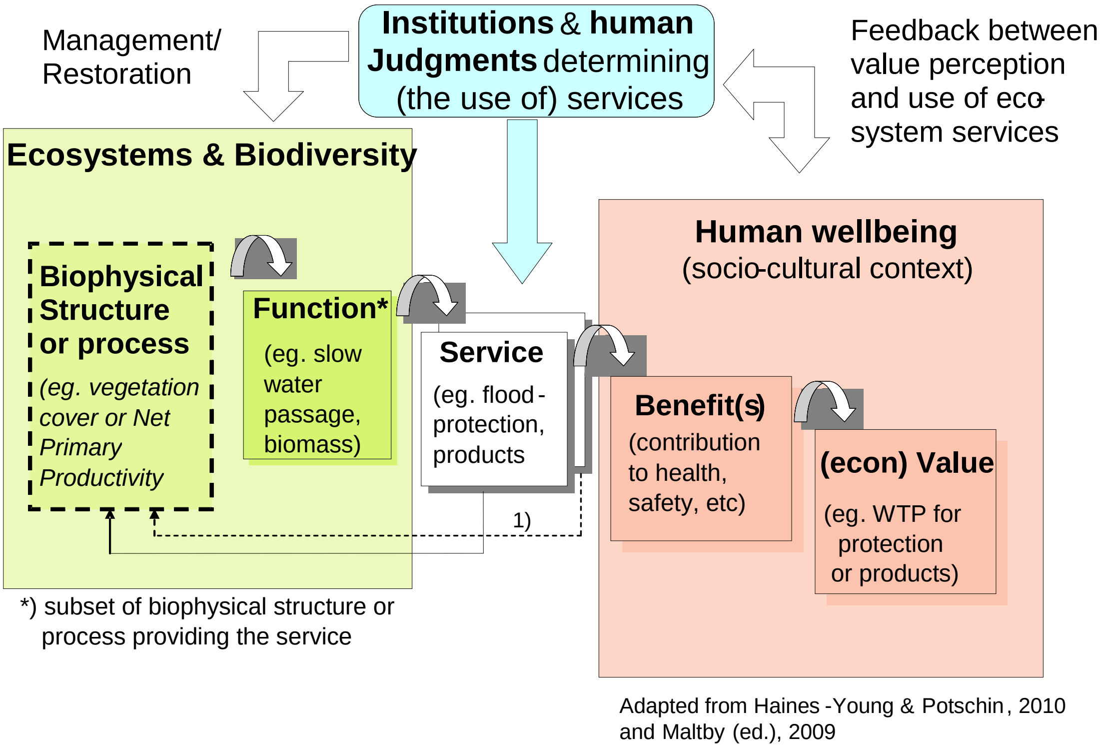

Figure 4: **The pathway from ecosystem structure and processes to human well-being**

2.3.1 From biophysical structure and process to ecosystem services and benefits

As Figure 4 shows, a lot goes on before services and benefits are provided, and decision-makers need to understand what this involves. It is therefore helpful to distinguish 'functions' from the even deeper ecological structures and processes in the sense that the functions represent the *potential* that ecosystems have to deliver a service which in turn depends on ecological structure and processes. For example, primary production (= process) is needed to maintain a viable fish population (= function) which can be used (harvested) to provide food (= service); nutrient cycling (=process) is needed for water purification (=function) to provide clean water (= provisioning service)[iv]. The benefits of these services are manifold, for example, food provides nutrition but also pleasure and sometimes even social identity (as part of cultural traditions); clean water can be used for drinking but also for swimming (pleasure) and other activities aimed at satisfying needs and wants. Thus, the role of woodlands in slowing the passage of water through a catchment is a function which has the potential of delivering a service (water flow regulation -> reduced flood risk) if some beneficiary exists to enjoy the benefit (safety).

Services are actually conceptualizations (‘labels’) of the “useful things” ecosystems “do” for people, directly *and* indirectly (see Glossary for the exact definition used in TEEB) whereby it should be realized that properties of ecological systems that people regard as ‘useful’ may change over time even if the ecological system itself remains in a relatively constant state.

Clearly delineating between ecological phenomena (functions), their direct and indirect contribution to human welfare (services), and the welfare gains they generate (benefits) is useful in avoiding the problem of double counting that may arise due to the fact that some services (in particular supporting and regulating services) are inputs to the production of others (Boyd and Banzhaf 2007; Wallace 2008; Fisher and Turner 2008; Balmford et al. 2008). Such differentiation is also crucial to provide a clear understanding of the spatial distribution of where the function occurs, where the provision of the service can be assessed, and ultimately where the benefits are appreciated. Although the distinction between functions, services and benefits is important, especially for economic valuation, it often is not possible to make a fully consistent classification, especially for regulating services (see section 3 for further discussion).

The short conclusion is that studies on ecosystem services should always be transparent on just what are considered services, and how they are being valued and measured. A critical missing link for some ecosystem services is the scant knowledge on how they are produced, maintained, and affected by system or abiotic changes and how they are related to levels of biodiversity. Information gaps will be rife throughout ecosystem service research and should always acknowledge the current uncertainty about how the ‘system’ works. It should also be realized that many people benefit from ecosystem services without realizing it, and thus fail to appreciate their value (importance). To make the dependence of human wellbeing on ecosystem services more clear, valuation studies should therefore not only include direct benefits (direct use values) but take due account of all the indirect benefits (indirect use values) and non-use values derived from ecosystem services.

Another issue is how to deal with potential benefits or the “likelihood of (future) use”, e.g. currently functions like wildlife (as potential food source), water purification (keeping rivers clean) or attractive scenery in a remote area may not be used but may have great (economic) potential for future use. Finally, it should also be recognized that ecosystems may provide *disservices*, for example, when they facilitate reproduction and dispersal of species that damage crops or human health[v]. In trade-off analysis, these disservices must be considered and, ultimately, the notion of benefits and ‘dis-benefits’ should be looked at within a consistent ecosystem accounting framework (e.g. EEA 2009).

2.3.2 From ecosystem services to (economic) value

Since the functioning of ecosystems and their services affect so many aspects of human welfare, a broad set of indicators can and should be used to measure the magnitude ('value') of their impact. As with the interpretation of the terms 'function', 'service' and 'benefits' (see above), much debate still surrounds the use of the term 'value' in assessing the benefits of ecosystems to human wellbeing. The Oxford English Dictionary defines value as "the worth, usefulness, importance of something". The Millennium Ecosystem Assessment defined value as "the contribution of an action or object to user-specified goals, objectives, or conditions" (after Farber et al. 2002), the measurement of which could include any kind of metric from the various scientific disciplines, e.g. ecology, sociology, economics (MA 2003).

In economics, 'value' is always associated with trade-offs, i.e. something only has (economic) value if we are willing to give up something to get or enjoy it. The common metric in economics is monetary valuation and some critics say the reliance on this metric has plagued many ecosystem service assessments, failing to incorporate several types of value which are critical to understanding the relationship between society and nature (e.g. Norgaard 1998; Wilson and Howarth 2002; MA 2005a; Christie et al. 2006). See also Box 1 and chapter 4 for further discussion.

In addition to economic valuation, other ways to analyze the importance of ecosystem services include livelihoods assessments, capabilities approaches that emphasize the opportunities available to people to make choices (e.g., Sen 1993), and vulnerability assessments. Such considerations are necessary for integrating into the analysis some dimensions of human well-being that cannot (or should not) be measured in terms of money, such as freedom of choice, human rights and intrinsic values. They are also important for measuring the services and benefits that are of cultural and philosophical (spiritual) nature. However, while monetary assessments only partially capture the total importance – i.e. value – of ecosystem services, they are vitally important for internalizing so-called externalities in economic accounting procedures and in policies that affect ecosystems, thereby influencing decision-making at all levels.

Box 1 Neoclassical Economics and its Discontents (by John Gowdy)

There is a long history of antagonism between traditional neoclassical economists and those advocating a more pluralistic approach to economic theory and policy. The debate has been less fruitful than it might have been because of the failure of many on both sides to be specific as to what is being criticized and defended.

Those of us who are critical of standard economic valuation methods need to be precise in what we are criticizing. The current debates raging in economics over the Stern Review, the current financial crisis, and the significance of the findings of behavioral economics, have shown that the problem with neoclassical economics is not valuation *per se* but with the assumptions of the core Walrasian model (named after the Swiss economist Leon Walras). The purpose of that model is to prove that competitive markets achieve Pareto efficiency, that is, no one can be made better off without making someone else worse off. This is called the First Fundamental Theorem of Welfare Economics. That proof does not work unless economic agents (firms and consumers) act independently of each other. That is, my economic decisions are based on self-regarding preferences and my decisions are in no way affected by how others think, behave or how much they have. Likewise, the production and pricing decisions of one firm are independent of the actions of other firms. The independence assumption has been falsified by thousands of empirical tests. It does not make good predictions of real economic behavior and offers a poor guide for economic policy. We need to replace “rational economic man” with a science-based model of human behavior, and the model of the perfectly competitive firm with one that includes competitive institutions, cultural norms, and biophysical transformations. The characterization of consumers and firms in the Walrasian model is driven by the mathematical requirements of constrained optimization and has little to do with real-world economic behavior.

But this doesn’t mean that markets are always inefficient or that prices have no meaning. It simply means that economic policy debates need to be decided on the basis of merit and evidence, not arbitrary equations or lines on a piece of paper. The effect of minimum wage laws is a good example. It’s easy to “prove” that they raise the unemployment rate by drawing a simple graph. But the real world evidence is much more complicated.

Economists frequently use a kind of “bait and switch” technique to justify their models. We begin with the proposition that “people do the best they can with the limited means at their disposal” which is reasonable. But in the Walrasian framework this becomes “People with well-defined, stable and self-regarding preferences maximize a smooth and continuous single-valued function that is twice differentiable… and so on.” Using the mathematical relationships in this model we can calculate shadow prices for natural resources, elasticities of substitution, total factor productivity etc. But it is not clear whether these estimates are based on empirical evidence or by the mathematical assumptions used to calculate them.

Regarding discounting future costs and benefits, we can reasonably say that individuals in general would rather have something now than in the distant future (but not always). But that is not the same as using a single precise discount rate number to value everything from biodiversity loss to the effects of climate change decades or even centuries in the future.

If we discard the straightjacket of Walrasian mathematics we can begin to sort out what can be priced, what can be measured without prices, and what cannot be measured at all but still valued.

3 TEEB-conceptual framework

Following the considerations discussed in section 2, and building upon the work done in the TEEB preparation phase, this section presents the Framework adopted for the TEEB study which also forms the "backbone" for the subsequent chapters in this report (Figure 5).

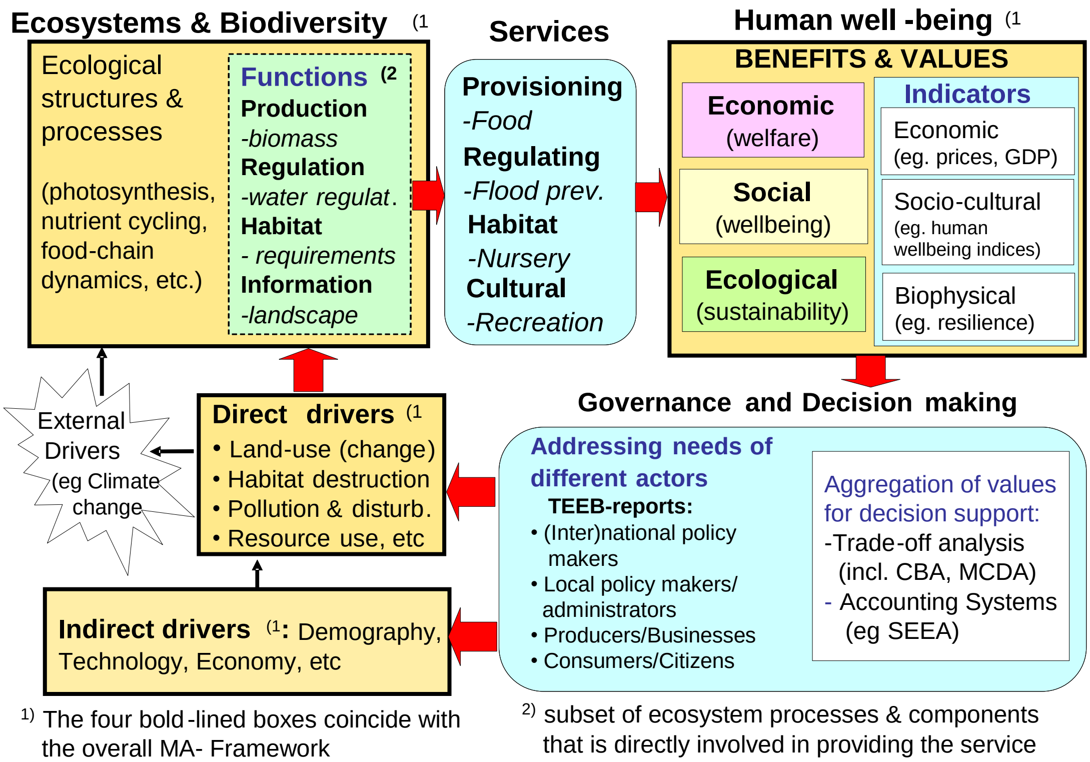

Figure 5 **Conceptual framework for linking ecosystems and human well-being**

To analyse the "Economics of Ecosystems and Biodiversity" a practical and consistent definition and typology of ecosystems (and biodiversity) is essential. In the TEEB assessment, we largely follow the definitions of the United Nations' 1992 Convention on Biological Diversity. Thus we see an *ecosystem* as the complex of living organisms and the abiotic environment with which they interact at a specified location. *Biodiversity* is the sum total of organisms including their genetic diversity and the way in which they fit together into communities and ecosystems (see Glossary for exact wording).

Based on various sources, TEEB proposes a typology of 12 main *biomes* (see Table 2), sub-divided into a much larger number of ecosystem types (see Appendix 1).

Table 2: Classification of main biomes in TEEB and remaining surface area

	Biome type (a	**Surface area remaining** (in 1000 km^2) (f			**Of which in more or less natural state (c**
		MODIS LC 2005 (b	Global Cover 2005 (b	Land Cover in 2000 (c	
1	**Marine / Open Ocean**	416,546			10-15 % (g
2	**Coastal systems**				40 % (h
3	**Wetlands**		1,799		? % (i
4	**Lakes & Rivers**		6,713		? % (i
5	**Forests**	28,936	46,652		
	Wooded tundra			2,596	93 %
	Boreal forest			17,611	85 %
	Cool coniferous forest			3,130	72 %
	Temperate mixed forest			5,914	49 %
	Temp. deciduous forest			4,718	43 %
	Warm mixed forest			5,835	52 %
	Tropical forest			9,149	76 %
6	**Woodland & shrubland**	35,829	9,766	7,870 8,773	67 % 44 %
	Mediterranean shrub (e			1,741	38 %
7	**Grass & Rangeland** *(d*	23,883	8,879	19,056	50 %
	Savanna only (e			15,604	57 %
8	**Desert**	18,154	34,753	22,174	83 %
9	**Tundra**		6,453	6,375	94 %
10	**Ice/Rock/Polar**	15,930	3,166	2,290	100 %
11	**Cultivated areas**	20,617	26,472		Not applicable
12	**Urban areas**	656	336		Not applicable

a) This classification is based on various sources (see Appendix 1, which also gives a more detailed list of ecosystems for each biome). The forest, woodland and grassland biomes are sub-divided here to accommodate data on the degree of human impact (last column) and this sub-division is therefore not completely identical with Appendix 1.

b) Data on surface area provided by Rosimeiry Portela, with help from Marc Steininger and Fabiano Godoy, all Conservation International.

c) Data provided by Leon Braat, based on work done by PBL/NEEA (Netherlands Environmental Assessment Agency) for COPI-I on terrestrial systems only (Braat et al., 2008). According to this source, the world total terrestrial area is 132,836,113 km^2 .

d) Including Steppe in PBL/NEEA data set (Braat et al. 2008).

e) These categories (Mediterranean shrub and savanna) are listed separately in the RIVM data set; their surface area is *not* included in sum-total for the respective main biome categories.

f) Data on surface areas differ substantially by source due to different interpretations of biome (land cover) classes.

g) Data on aquatic systems is more difficult to find and interpret than for terrestrial systems, but Halpern et al. (2008) estimated that at least 40% of ocean systems are medium - very highly affected by human impact and the other 60% face low – medium impact. Only 10-15% can be considered to be in a more or less pristine state (Halpern personal communication, August 2009).

h) For coastal systems the figures are even harder to estimate, but according to Halpern et al. (2009), 60 % of the global coastline experience low to high impact from land-based human activities. This estimate was based on four of the most pervasive land-based impacts on coastal ecosystems: nutrient input; organic and inorganic pollution; and the direct impact of coastal human populations. If we add ocean-based human impact (i.e. over-fishing, pollution from ships, etc.) this figure will surely be much higher: a map published by UNEP (2006) shows that only about 40% of the coasts face "little or no" impact from human actions.

i) For wetlands, rivers and lakes, no reliable (global) data were found in this phase of the TEEB study but it will be attempted to further verify and complete this table in the coming months.

As Table 1 indicates, most of the biomes and associated ecosystems have been converted to human-dominated systems (agriculture, aquaculture, plantations, etc.) to a greater or lesser extent (on average approximately 1/3 of the area) or are otherwise affected by human activity, e.g. through over-exploitation and pollution of marine systems (Nelleman et al. 2008) and damming of rivers. Thus, 'biomes' or other ecology-based land categories have actually become a "construct" (e.g., Ellis 2008, Kareiva et al. 2007). Because of the predominance of so-called 'socio-ecological mosaics' (i.e. a patchwork of landscape units that range from intensively managed to unmanaged areas, all within the same landscape), fine-grained spatial analysis should be a core of any Ecosystem Service assessment (see Box 2).

Box 2.: Spatial explicitness and scale

A major critique of early ecosystem service valuation work was the rudimentary treatment of ecological systems at the scale of biomes, and the extrapolation of site specific values across the entire globe (Bockstael et al. 2000; Naidoo et al. 2008). At the other end of the spectrum, the utility of plot-scale experiments for policy formation is questionable. Recognizing this trade-off, advanced research in ecosystem services has focused on spatially-explicit economic and ecological models, moving away from standard lookup tables assuming constant marginal values and utilizing benefit transfer based on ecosystem type (Barbier et al. 2008; Nelson et al. 2009; Naidoo and Ricketts 2006; Polasky et al. 2008; Bateman et al. 2003).

Working at both ecologically understandable and policy relevant scales also allows researchers to more fully understand the values and perceptions of the relevant stakeholders (Hein et al. 2006; Fisher et al. 2008; Granek et al. 2009). Another major reason why spatial explicitness is important is that the production and use of services from ecosystems vary spatially, as along with the economic benefits they generate (many of which are local in nature), and of course the costs of action, so it matters to human well-being where conservation actions are implemented. A spatially explicit assessment of the impacts of action and quantification of benefits and costs is also helpful to show the possible mismatch between the ecological and the socio-economic scales of decision making, service provision and use, and between winners and losers in different scenarios. It is thus essential for designing effective and equitable policy interventions. (Balmford et al. 2008).

In trade-off analysis of land use change, ideally the costs and benefits of the transitions, and all or at least the main intermediate states (see Figure 6), should be based on the economic value of the total bundle of services provided by each transition or management state. This level of detail is impossible within this phase of the TEEB assessment, both for time limitations and given the paucity of studies that compare the provision of services by an ecosystem under alternative management states (Balmford et al. 2002; ICSU-UNESCO-UNU 2008), but it should be a high priority subject for follow-up studies.

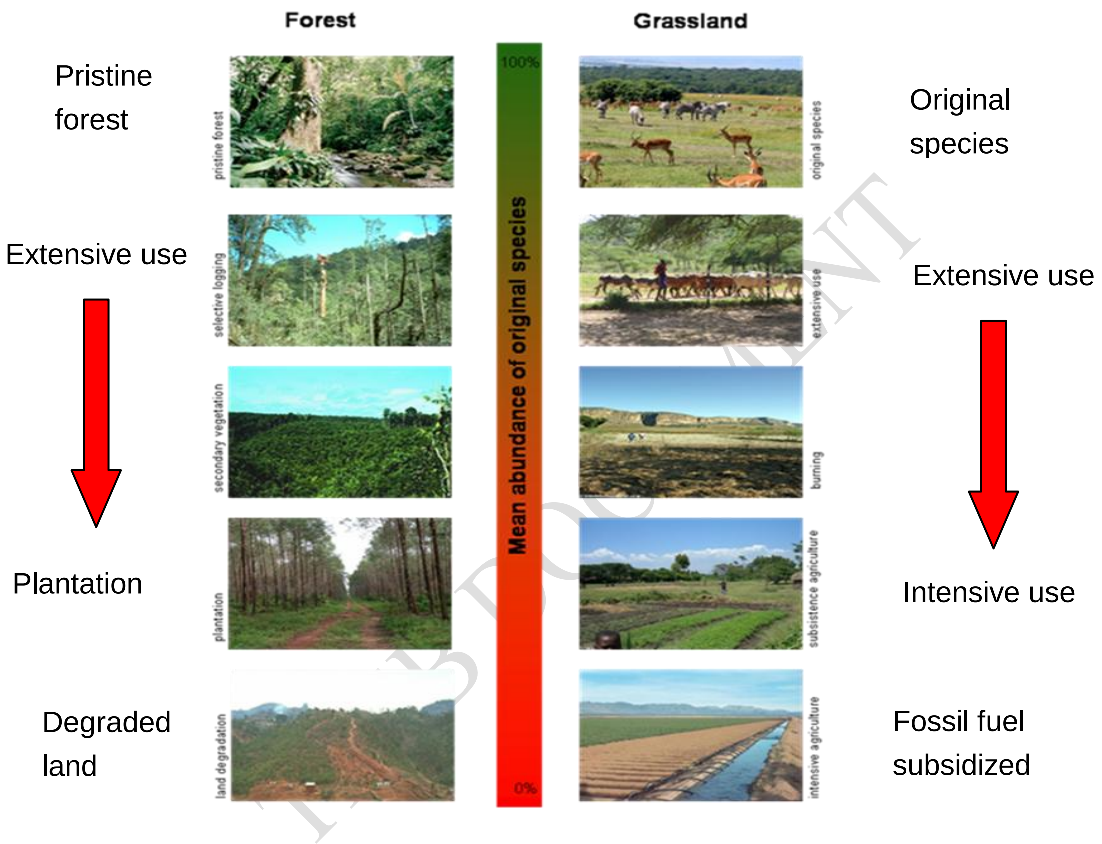

Figure 6 **Two examples of degradation pathways showing transition phases between natural and human-dominated (eco)systems**
Source: Braat et al. (2008).

The recognition of tangible ecological or physical boundaries of ecosystems, however arbitrary it may sometimes be, provides an important basis for adaptive and practical management through the mapping of particular functions and landscape units, or even so-called ‘service-providing units’ (see chapter 2).

3.1 Ecosystem structure, processes and functions

The TEEB framework (Figure 5) starts with the upper-left hand box which distinguishes ecosystem structure, processes and functions. *Ecosystem functions* are defined as a subset of the interactions between ecosystem structure and processes that underpin the capacity of an ecosystem to provide goods and services. The building blocks of ecosystem functions are the interactions between structure and processes, which may be physical (e.g. infiltration of water, sediment movement), chemical (e.g. reduction, oxidation) or biological (e.g. photosynthesis and denitrification), whereby 'biodiversity' is more or less involved in all of them, although the precise detail of the relationship is often unclear or limited (see chapter 2).

The fundamental challenge is the extent to which it is practical (possible?) to fully predict the actual functioning of any defined ecosystem unit when relatively few (and rarely replicated) studies worldwide are available. It is often necessary to rely on various combinations of seemingly-appropriate indicators of ecosystem condition and function (see chapter 3) which can in theory be applied more generally than in just individual cases.

3.2 Typology of ecosystem services

Ecosystem services are defined in TEEB as "the direct and indirect contributions of ecosystems to human well-being." This basically follows the MA-definition except that it makes a finer distinction between services and benefits and explicitly acknowledges that services can benefit people in multiple and indirect ways (see section 3 for a more detailed discussion).

Based on the TEEB preparatory phase and other assessments and meta-analysis (see section 2), TEEB proposes a typology of 22 ecosystem services divided in 4 main categories; provisioning, regulating, habitat and cultural & amenity services, mainly following the MA classification (see Table 3 and Appendix 2 for a more detailed list and comparison with the main literature).

An important difference we adopt here, as compared to the MA, is the omission of Supporting Services such as nutrient cycling and food-chain dynamics, which are seen in TEEB as a subset of ecological processes. Instead, the Habitat Service has been identified as a separate category to highlight the importance of ecosystems to provide habitat for migratory species (e.g. as nurseries) and gene-pool "protectors" (e.g. natural habitats allowing natural selection processes to maintain the vitality of the gene pool). The availability of these services is directly dependent on the state of the habitat (habitat requirements) providing the service. In case commercial species are involved, such as fish and shrimp species that spawn in mangrove systems (= nursery service) but of which the adults are caught far away, this service has an economic (monetary) value in its own right. Also the importance of the gene-pool protection service of ecosystems is increasingly recognized, both as "hot spots" for conservation (in which money is increasingly invested) and to maintain the original gene-pool of commercial species (which we are increasingly being imitated through the creation of botanic gardens, zoos and gene banks).

Before economic valuation can be applied, the performance or availability of ecosystem services has to be measured in biophysical terms (see Chapters 2 and 3). In some cases the state of ecological knowledge and the data availability allow using some direct measures of services, while in other cases it is necessary to make use of proxies.

Table 3 Typology of ecosystem services in TEEB

	Main service types
	PROVISIONING SERVICES
1	**Food** (e.g. fish, game, fruit)
2	**Water** (e.g. for drinking, irrigation, cooling)
3	**Raw Materials** (e.g. fiber, timber, fuel wood, fodder, fertilizer)
4	**Genetic resources** (e.g. for crop-improvement and medicinal purposes)
5	**Medicinal resources** (e.g. biochemical products, models & test-organisms)
6	**Ornamental resources** (e.g. artisan work, décorative plants, pet animals, fashion)
	REGULATING SERVICES
7	**Air quality regulation** (e.g. capturing (fine)dust, chemicals, etc)
8	**Climate regulation** (incl. C-sequestration, influence of vegetation on rainfall, etc.)
9	**Moderation of extreme events** (eg. storm protection and flood prevention)
10	**Regulation of water flows** (e.g. natural drainage, irrigation and drought prevention)
11	**Waste treatment** (especially water purification)
12	**Erosion prevention**
13	**Maintenance of soil fertility** (incl. soil formation)
14	**Pollination**
15	**Biological control** (e.g. seed dispersal, pest and disease control)
	HABITAT SERVICES
16	**Maintenance of life cycles of migratory species** (incl. nursery service)
17	**Maintenance of genetic diversity** (especially in gene pool protection)
	CULTURAL & AMENITY SERVICES
18	**Aesthetic information**
19	**Opportunities for recreation & tourism**
20	**Inspiration for culture, art and design**
21	**Spiritual experience**
22	**Information for cognitive development**

Source: based on/adapted (mainly) from Costanza et al. (1997), De Groot et al. (2002), MA (2005a), Daily, Ehrlich, Mooney, et al. (2008). See Appendix 2 for details.

Actual measurements of ecosystem services should be split into a) the capacity of an ecosystem to provide a service (e.g., how much fish can a lake provide on a sustainable basis), and b) the actual use of that service (e.g., fish harvesting for food or for use in industrial processing). Measurement of the importance (value) of that fish in terms of nutrition value, a source of income and/or way of life is then part of the "human value domain".

When applying valuation, it is necessary to clearly distinguish between potential and actual use of services with direct use value (notably provisioning and some cultural services), and services that have indirect use (notably regulating, habitat and some services). Since most ecosystems provide a bundle of services, and the use of one service often affects the availability of other services, (economic) valuation should consider not only (marginal) values from the flows of individual services but also take due account of the "stock value" (i.e. the entire ecosystem) providing the total bundle of services.[vi] When applying economic valuation the actual management regime of the ecosystem (which is determined by the institutional arrangements) should be taken into account. This regime will influence the expected value of future flows of services, which will differ depending on whether it leads to sustainable or unsustainable uses (Mäler 2008).

3.3 Human well-being: typology of benefits and values

Following the MA approach, the TEEB framework (Figure 5) makes a distinction between ecological, socio-cultural and economic benefits and values. The reason for separating benefits and values is because people have needs which, when fulfilled, are translated into (more or less objectively measurable) benefits. For example, catching fish from the ocean gives us food (health), but also cultural identity (as a fisherman/-woman) and income. How we value these benefits is subjective: some people will value the income much higher than their cultural identity (social ties etc) and may be willing to give up one aspect of their wellbeing (cultural identity) over another (e.g. material wealth). Thus, different values can be attached to a particular benefit.

Although the TEEB study focuses primarily on the measurement of economic values and the assessment of costs and benefits in a welfare economics approach, it includes equity considerations in particular for the aggregation of benefits over time and over groups of people. It specifically analyses the relationships between ecosystems and poverty ('GDP of the poor'), because of the higher dependence of the poor on ecosystem services for their livelihood (TEEB 2008).

Of course, it should also be acknowledged that many native communities ('ecosystem people') still entirely, and directly, depend on ecosystems and their services for their survival, as well as the importance of ecosystems for providing people with the ability to choose certain ways of life that they may value.

The three main types of benefits (well-being aspects) and related values and valuation metrics are briefly introduced below (for more detailed information, see chapter 3 (biophysical indicators – linked to ecological "values"), chapter 4 (socio-cultural values) and chapter 5 (economic values).

3.3.1 Ecological benefits and values

The ecological importance (value) of ecosystems has been articulated by natural scientists in reference to the causal relationships between parts of a system such as, for example, the value of a particular tree species to control erosion, or the value of one species to the survival of another species or of an entire ecosystem (Farber et al. 2002). At a global scale, different ecosystems and their constituent species play different roles in the maintenance of essential life-support processes (such as energy conversion, biogeochemical cycling, and evolution) (MA 2003).

Ecological measures of value (importance) are, for example, integrity, 'health', or resilience, which are important indicators to determine critical thresholds and minimum requirements for ecosystem service provision. These measures of value should be distinguished from what can be included in economic values because although they contribute to welfare, they cannot readily be taken into account in the expression of individual preferences, as they are too indirect and complex, albeit they may be critical for human survival. The related value paradigm could be formulated as the importance people attach to a healthy, ecologically stable environment, both as a contribution to human survival (instrumental value) and for intrinsic reasons (values). Although the notion of ecological value is still much debated, the 'value' of natural ecosystems and their components should be recognized in terms of their contribution to maintaining life on earth, including human survival in its own right (Farber et al. 2002).

3.3.2 Socio-cultural benefits and values

For many people, biodiversity and natural ecosystems are a crucial source of non-material well-being through their influence on mental health and their, historical, national, ethical, religious, and spiritual values. While conceptual and methodological developments in economic valuation have aimed at covering a broad range of values, including intangible ones (see the concept of Total Economic Value below), it can be argued that socio-cultural values cannot be fully captured by economic valuation techniques (cf. chapters 4 and 5) and have to be complemented by other approaches in order to inform decision-making. This is notably the case where some ecosystem services are considered essential to a people's very identity and existence. To obtain at least a minimum (baseline) measure of importance of socio-cultural benefits and values several metrics have been developed such as the Human Wellbeing Index.

3.3.3 Economic benefits and values

Biodiversity and ecosystem services are important to humans for many reasons. In economic terms, this can be considered as contributing to different elements of 'Total Economic Value', which comprises both use values (including direct use such as resource use, recreation, and indirect use from regulating services) and non-use values, e.g. the value people place on protecting nature for future use (option values) or for ethical reasons (bequest and existence values). The economic importance of most of these values can be measured in monetary terms, with varying degrees of accuracy, using various techniques (including market pricing, shadow pricing and questionnaire based). Chapter 5 gives a detailed overview of economic values and monetary valuation techniques.

3.4 Governance and decision making

In making decisions at any level (private, corporate or government), decision-makers are faced with the dilemma of how to balance (weigh) ecological, socio-cultural and economic values. Preferably, the importance of each of these value-components should be weighted on its own (qualitative and quantitative) dimension, e.g. through Multi-Criteria Decision Analysis. However, since TEEB is focusing on the economic, notably monetary, consequences of the loss of biodiversity, concentrates TEEB on aggregation *(1)* and economic trade-off issues *(2).* To make the link with standard macroeconomic indicators, the role of ecosystem services in environmental-economic accounting *(3)* should also be mentioned as a promising field of analysis to inform economic decisions (EEA 2009). Finally, awareness raising and positive incentives *(4)* are essential tools for better decision-making.

(1) Aggregating monetary values

Aggregation involves bringing together all the information on the monetary values of ecosystem services by ecosystem type into a single matrix to attain an aggregate monetary value of all delivered ecosystem services. This is the task of Chapter 7. Effective aggregation is challenging. Key issues requiring consideration include:

- *Accounting for uncertainties in the monetary valuation of individual services,* including possible biases due to the use of different valuation methods (see chapter 5 for discussion).
- *Interdependencies between ecosystem services at the ecosystem scale*, including issues of double counting, competing services, bundled services, etc.
- *Aggregation of values over individuals and groups of people*. The relative importance of ecosystem services will vary between different groups of people, e.g., regarding income level or dependence on ecosystem services. To integrate such considerations some adjustments can be applied such as equity weightings (Anthoff et al. 2009).
- *Aggregation of values over spatial scales*. Different ecosystem services may be best considered at different spatial scales. For example, water regulation is best considered at a watershed scale, while carbon sequestration can be considered on a national or global scale. Aggregation should take these differences into account.
- *Aggregation of values over time*: Protecting biodiversity today may have costs and benefits to future generations. In economics, discounting is a common practice to compare these future costs and benefits with current values. An important issue is the selection of the most appropriate discount rate in different decision-making contexts. Chapter 6 will further explore these issues.

(2) Trade-off analysis

A trade-off occurs when the extraction of an ecosystem service has a negative impact on the provision of other services. For example, timber extraction from a forest will affect, among others, vegetation structure and composition, visual quality and water quality which will preclude or at least affect the continuous provision of other services (e.g. wildlife harvesting, carbon sequestration, recreation) over time, since loss of structure implies loss of function, and consequently of other services and their derived benefits. Approaches to trade-off analysis include: multi-criteria (decision) analysis, cost-benefit analysis and cost-effectiveness analysis.

The foundational strength of *Cost Benefit Analysis (CBA)* is finding the 'net' benefit of an activity. Since the costs and benefits of an activity (or scenario) have different functional relationship in different circumstances – utilizing a 'benefits only' approach could greatly mislead decision-making (Naidoo et al. 2006). This benefits-only approach was common in early ecosystem service assessments (Balmford et al. 2002). A notable early exception is research on the fynbos in South Africa, where researchers enumerated the benefits and costs of both an invasive species eradication campaign and a do-nothing approach (Van Wilgen et al. 1996). An understanding of costs is also crucial in ecosystem service research since the complexity of benefit delivery might preclude a full understanding of service delivery. In these cases a *cost-effectiveness approach* can be highly informative especially where the costs vary more than the benefits (Ando et al. 1998; Balmford et al. 2003; Naidoo et al. 2006, EEA 2009).

(3) Systems of Ecological–Economic Accounting: macro-economic implications

A growing number of governments recognize the need to include ecosystem services in economic accounts in order to ensure that their contribution to well-being is recorded in the macroeconomic indicators that are the most widely acknowledged and used in policy making. If ecosystems are regarded as assets that provide services to people, then accounts can be used to describe the way they change over time in terms of stocks and flows. These changes can be described both in physical terms, using various indicators of ecosystem quantity and quality, and ultimately in monetary values (EEA, 2009). Ecosystem accounting, linked to geographical information systems and to socio-economic data, can thus offer a useful framework for systematically collecting and analyzing data to support assessments of changes in the production and use of ecosystem services, taking into account their spatial heterogeneity.

Several relevant initiatives are currently under way. For example, the European Environment Agency is developing a framework for land and ecosystem accounts for Europe, building on land cover data and following the System of Environmental and Economic Accounts (SEEA) guidelines of the United Nations.

The development of ecosystem accounting will have to be gradual, integrating progressively more ecosystem services, and build on existing information in different countries. This is addressed in more detail in the TEEB report for national and international policy makers (TEEB 2009). An analysis of how the value of some ecosystem services can be recorded at macroeconomic level for some economic sectors is presented in chapter 9 of this report.

(4) Awareness raising and positive incentives

Of special importance in the TEEB context are the numerous decisions by producers and consumers affecting ecosystems (TEEB reports for business and citizens), and the policy changes necessary to ensure that decisions taken at various governmental levels (TEEB 2009 and TEEB for administrators), do not lead to greater degradation of ecosystems and even improve their condition (see section 3.6).

An important step towards the conservation and sustainable use of biodiversity and ecosystem services lies in accounting for the positive and negative externalities associated with human activities. Rewarding the benefits of conservation through *payments for environmental services* (e.g., Landell-Mills and Porras 2002; Wunder 2005) or ecological fiscal transfers (Ring 2008) is as important as the realignment of perverse subsidies that all too often incentivize unsustainable behaviors (TEEB 2009, chapters 5 and 6).

A growing societal awareness of the need for research and development, and for changes in policy, practice and law, can help us pursue sustainable ecosystem management and resource use, and engage in eco-regional planning and large-scale restoration and rehabilitation of renewable and cultivated natural capital (Aronson et al. 2007).

3.5 Scenarios and drivers of change

Efforts aimed at changing behavior towards, and impact on ecosystems and biodiversity must take into account that ecosystems have always been dynamic, both internally and in response to changing environments.

The importance of using scenarios in ecosystem service assessments is beginning to be realized as early assessments presented a static picture in a rapidly changing world. The necessity of providing counter-factuals is now being demanded of conservation research (Ferraro and Pattanayak 2006) and should become the norm in ecosystem service research as well. The generation of scenarios is particularly important for monetary valuation, since scenarios enable analysis of changes in service delivery which are required to obtain marginal values. Making an analysis in incremental terms avoids (or at least reduces) the methodological difficulties, which vary depending on the magnitude of the changes. Such difficulties arise when attempting to estimate total values, related to the non-constancy of marginal values associated with the complete loss of an ecosystem service. Such approaches are also in general more relevant for decision-making in real-life circumstances.

In the TEEB context, comparing the outputs under several scenarios will inform decision makers of the welfare gains and losses of alternative possible futures and different associated policy packages. This is also important for non-monetary valuation changes, but more from a social understanding aspect than for analytical robustness. For each scenario to be elaborated, we must analyze the likely consequences of drivers that directly affect the status, current management and future trajectories of ecosystems and biodiversity (and thus of the services and values they represent).

Indirect drivers of ecosystem change include demographic shifts, technology innovations, economic development, legal and institutional frameworks, including policy instruments, the steady loss of traditional knowledge and cultural diversity and many other factors that influence our collective decisions (OECD 2003; MA 2005b; OECD 2008). These (indirect) drivers affect the way people directly use and manage ecosystems and their services.

Direct drivers can be organized in negative, neutral and positive categories. *Negative drivers* include, among others, habitat destruction, over-use of resources such as largely unrestrained overfishing of the oceans of the world and pollution (leading among others to climate change). Examples of *neutral drivers* would be land use change (which can have positive or negative consequences for ecosystems and biodiversity, depending on the context and management regime). Increasing intensification and industrialization of agriculture and animal husbandry should also be placed in a broader context: intensification (provided it is done in a sustainable manner), can provide extra space for natural habitat. Finally, *positive drivers* for enhancing natural capital would include ecosystem conservation and restoration, development of sustainable management regimes and use of environmental-friendly technologies, aimed at reducing human pressure on ecosystems and biodiversity (e.g. organic farming, eco-tourism, renewable energy, etc). Clearly, even 'positive drivers' can have negative impacts on ecosystems and biodiversity, when applied in the wrong place or context, so the effects of any direct driver on ecosystems need to be carefully analyzed through the TEEB framework.

3.6 Linking ecosystem service values to decision-making: the TEEB guidance reports

TEEB brings together state-of-the art research on assessing and valuing ecosystem services to help policy makers, local authorities, companies and individuals in making decisions with respect to their responsibilities in safeguarding biodiversity. Decision-makers at different organizational levels, both public and private, affect drivers of ecosystem change such as demographic, economic, socio-political, scientific and technological as well as cultural and religious drivers, which in turn affect ecosystem services and human wellbeing. Building on a more refined valuation framework and methodology that is more suitable for capturing economic values and policy-relevant information, TEEB will develop specific guidance documents or “deliverables” addressing decision-making at different levels in different contexts by different actors.

The first guidance document addresses *policy makers* (TEEB 2009). It explores the consequences of international and national policies on biodiversity and ecosystems and presents a TEEB policy toolkit for decision-makers at various governmental levels. By demonstrating the value attached to ecosystem services and considering them in concrete policies, instruments and measures (e.g., subsidies and incentives, environmental liability, market creation, national income accounting standards, trading rules, reporting requirements, eco-labelling), it aims to enhance biodiversity and ecosystem protection as a prerequisite for maintaining natural service levels.

Local administrators are addressed by the second TEEB deliverable. It incorporates values of ecosystem services in location-specific, cost-benefit and cost-effectiveness analysis, and their use in methods and guidelines for implementing payments for ecosystem services, as well as equitable access and benefit-sharing arrangements for genetic resources and protected areas.

The third TEEB deliverable focuses on the *business* end-user. It aims to provide a framework for assessing the business impacts on biodiversity and ecosystems, both for measuring and managing risks and identifying and grasping new market opportunities for private enterprises.

Last, but not least, *individuals and consumer organizations* are addressed by the fourth TEEB deliverable. It covers how to reduce their impacts on wild nature while influencing producers through private purchasing decisions. This will include steps to improve consumer information on the land, water and energy resources used in producing foods and consumer goods.

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[i] To avoid having to use both the terms ecosystems and biodiversity simultaneously all the time, the term ‘ecosystem’ is used to include ‘biodiversity’ throughout the chapter unless indicated otherwise (see Glossary for further explanation of these terms, and chapter 2 for a more in-depth discussion).

[ii] All key terms used in TEEB are included in the Glossary, usually indicated in italics in the text.

[iii] Acknowledging that it is quite impossible to mention all who contributed to the development of the concept of ecosystem services, some key authors and initiatives are listed in Annex 5 in the “Scoping the Science” report by Balmford et al. (2008).

[iv] Note that water purification is also listed as a regulating service in Appendix 2, in case the benefit is related to waste treatment. As mentioned at the beginning of section 2.3 a fully unambiguous classification system probably does not exist because the mix of ecosystem structure – process – function that provides the service changes depending on the benefit pursued.

[v] It should also be realized that many of these disservices are the result of bad planning or management and thus often man-made. For example “normalizing” rivers (leading to floods), cutting forest on hill slopes (causing erosion and landslides), and disturbing natural food webs (leading to outbreaks of pests).

[vi] In this context the ecosystem can be seen as the ‘factory’ providing (a bundle of) services. It is normal that, for example, car factories include the costs of maintaining the machines and buildings in the price of the car but for

timber or fish coming from a forest or lake we usually exclude the maintenance costs of the natural capital (stock) providing the service is usually excluded.

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Appendices

1. Classification of ecosystems used in TEEB

2. Ecosystem service classification: brief literature survey and TEEB classification

3. How the TEEB framework can be applied: the Amazon case

Appendix 1: Classification of ecosystems used in TEEB

	LEVEL 1 (Biomes)		**LEVEL 2 (ecosystems)**
1	**Marine / Open ocean**	**1.0**	**Marine / Open ocean**
		1.1	Open ocean
		1.2	Coral reefs (*, (#
2	**Coastal systems**	**2.0**	**Coastal systems (excluding wetlands)**
		2.1	- Seagrass/algae beds
		2.2	- Shelf sea
		2.3	- Estuaries
		2.3	- Shores (rocky & beaches)
3	**Wetlands**	**3.0**	**Wetlands – general (coastal & inland)**
			(coastal wetlands)
		3.1	- Tidal Marsh (coastal wetlands)
		3.2	- Mangroves (#
			(Inland wetlands)
		3.3	- Floodplains (incl. swamps/marsh)
		3.4	- Peat-wetlands (bogs, fens, etc.)
4	**Lakes/Rivers**	**4.0**	**Lakes/Rivers**
		4.1	- Lakes
		4.2	- Rivers
5	**Forests**	**5.0**	**Forests – all**
			(Tropical Forest)
		5.1	- Tropical rain forest (#
		5.2	- Tropical dry forest
			(Temperate forests)
		5.3	- Temperate rain/Evergreen
		5.4	- Temperate deciduous forests
		5.5	- Boreal/Coniferous forest
6	**Woodland & shrubland**	**6.0**	**Woodland & shrubland** (“dryland”)
		6.1	- Heathland
		6.2	- Mediterranean scrub
		6.3	- Various scrubland
7	**Grass/Rangeland**	**7.0**	**Grass/Rangeland**
		7.1	- Savanna etc
8	**Desert**	**8.0**	**Desert**
		8.1	- Semi-desert
		8.2	- True desert (sand/rock)
9	**Tundra**	**9.0**	**Tundra**
10	**Ice/Rock/Polar**	**10.0**	**Ice/Rock/Polar**
11	**Cultivated**	**11.0**	**Cultivated**
		11.1	Cropland (arable land, pastures, etc.)
		11.2	Plantations / orchards / agro-forestry, etc.
		11.3	Aquaculture / rice paddies, etc.
12	**Urban**	**12.0**	**Urban**

Source: Based on mix of classifications, mainly MA (2005a) and Costanza et al. (1997) which in turn are based on classifications from US Geol. Survey, IUCN, WWF, UNEP and FAO.
*) usually placed under “coastal” but it is proposed to put this under “marine”.
#) These three ecosystems are dealt with separately in the monetary valuation (chapter 7).

Appendix 2: Ecosystem service classification: brief literature survey and TEEB classification

Various sources (1	Millennium Ecosystem Assessment (2005a)	Daily et al. (2008)		**TEEB classification**
PROVISIONING	PROVISIONING			**PROVISIONING**
Food (fish, game, fruit)	Food	Seafood, game	1	Food
Water availability [RS] (2	Fresh water		2	Water (2
Raw materials (e.g. wood)	Fibre	Timber, fibers	3	Raw materials
Fuel & energy (fuel-wood, organic matter, etc.)	,, ?	Biomass fuels		
Fodder & fertilizer	,, ?	Forage		
Useful genetic material,	Genetic resources	- industrial products	4	Genetic resources
Drugs & pharmaceutical	Biochemicals	Pharmaceuticals	5	Medicinal resources
Models & test organisms	- ?	- industrial products		
Resources for fashion, handicraft, decorative, etc.	Ornamental resources	- ?	6	Ornamental resources
REGULATING	REGULATING			**REGULATING**
Gas regulation/air quality	Air quality regulation	Air purification	7	Air purification
Favorable climate (incl. C-sequestration)	Climate regulation	Climate stabilization	8	Climate regulation (incl. C-sequestration)
Storm protection	- ?	Moder. of extremes	9	Disturbance prevention or moderation
Flood prevention	Water regulation	Flood mitigation		
Drainage & natural irrigation (drought prevent.)	‘ ’	Drought mitigation	10	Regulation of water flows
Clean water (waste treatment)	‘ ’	Water purification	11	Waste treatment (esp. water purification)
Erosion prevention	Erosion regulation	Erosion protection	12	Erosion prevention
Maintenance of productive and “clean” soils	Soil formation [supporting service]	Soil generation and preservation	13	Maintaining soil fertility
Pollination	Pollination	Pollination	14	Pollination
(biol. control)		Seed dispersal	15	Biological control
Pest & disease control	Pest regulation	Pest control		
	Human disease regulat.			
HABITAT/SUPPORT	**SUPPORTING**	*(3*		**HABITAT**
Nursery-service	e.g. Photosynthesis, primary production, nutrient cycling		16	Lifecycle maintenance
Maintenance of biodiversity		Maintenance of biodiversity	17	Gene pool protection
CULTURAL (& Amenit.)	**CULTURAL**			**CULTURAL** & Amenity
Appreciated scenery (incl. tranquility)	Aesthetic values	Aesthetic beauty	18	Aesthetic information
Recreation & tourism	Recreat. & eco-tourism		19	Recreation & tourism
Inspiration for art etc.	- ?		20	Inspiration for culture, art and design
Cultural heritage	Cultural diversity			
Spiritual & religious use	Spirit. & religious val.		21	Spiritual experience
Use in science & education	Knowledge systems Educational values	Intellectual stimulation	22	Information for cognitive development

1) Mainly based on/adapted from Costanza et al. (1997) and De Groot et al. (2002).
2) Water is often placed under Regulating Services [RS] but in TEEB the consumptive use of water is placed under provisioning services.
3) Daily et al. (2008) do not use main categories and also included detoxification and decomposition of waste, nutrient cycling, and UVb-protection as services.

Chapter 2

Biodiversity, ecosystems and ecosystem services

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Contents

Key Messages

- All ecosystems are shaped by people, directly or indirectly and all people, rich or poor, rural or urban, depend on the capacity of ecosystems to generate essential ecosystem services. In this sense, people and ecosystems are interdependent social-ecological systems.

- The ecosystem concept describes the interrelationships between living organisms (people included) and the non-living environment and provides a holistic approach to understanding the generation of services from an environment that both delivers benefits to and imposes costs on people.

- Variation in biological diversity relates to the operations of ecosystems in at least three ways:
  1. increase in diversity often leads to an increase in productivity due to complementary traits among species for resource use, and productivity itself underpins many ecosystem services,
  2. increased diversity leads to an increase in response diversity (range of traits related to how species within the same functional group respond to environmental drivers) resulting in less variability in functioning over time as environment changes,
  3. idiosyncratic effects due to keystone species properties and unique trait-combinations which may result in a disproportional effect of losing one particular species compared to the effect of losing individual species at random.

- Ecosystems produce multiple services and these interact in complex ways, different services being interlinked, both negatively and positively. Delivery of many services will therefore vary in a correlated manner, but when an ecosystem is managed principally for the delivery of a single service (e.g. food production), other services are nearly always affected negatively.

- Ecosystems vary in their ability to buffer and adapt to both natural and anthropogenic changes as well as recover after changes (i.e. resilience). When subjected to severe change, ecosystems may cross thresholds and move into different and often less desirable ecological states or trajectories. A major challenge is how to design ecosystem management in ways that maintain resilience and avoids passing undesirable thresholds.

- There is clear evidence for a central role of biodiversity in the delivery of some – but not all - services, viewed individually. However, ecosystems need to be managed to deliver multiple services to sustain human well-being and also managed at the level of landscapes and seascapes in ways that avoid the passing of dangerous tipping-points. We can state with high certainty that maintaining functioning ecosystems capable of delivering multiple services requires a general approach to sustaining biodiversity, in the long-term also when a single service is the focus.

1 Introduction

This chapter explores current understanding of the relationships between biodiversity, the structure and functioning of ecosystems, and the provision of ecosystem services. It aims specifically to clarify:

- The nature of and evidence for the links between biodiversity, ecosystems, and ecosystem services;
- Ecosystem responses to anthropogenic impacts;
- The risks and uncertainties inherent in management of ecosystems that developed long before the evolution of *Homo sapiens*.

A basic level of understanding is an essential prerequisite to the appropriate application of economic analysis. This chapter highlights the complexities of the concepts of biodiversity and ecosystems, and examines the relationships between biodiversity, ecosystem functioning and ecosystem services. The interactions among the various assemblages of biotic and abiotic components into ecosystems are assessed based on our current scientific knowledge. This evidence is further discussed in the context of how to help inform the policy agenda on the connections between biodiversity and ecosystem services.

The chapter gives a review of the individual ecosystem services themselves with commentary and analysis on the important factors underpinning the services, gaps in knowledge and uncertainties. Recognizing that in reality, ecosystems generate multiple services, this chapter examines the complications arising from 'bundles' of ecosystem services, where strategic priorities may result in trade-offs in service provision. The need for practical approaches to the recognition, quantification and mapping of ecosystem services is examined, and a synthesis presented of the alteration of biodiversity and ecosystems and their functioning with increasing known impacts of global change. Analysis of the growing biophysical knowledge base is essential to help economists understand and interpret the dynamics and complex interactions among living organisms, the abiotic environment and diverse cultural and socio-economic contexts.

1 Biodiversity and ecosystems

1.1 Theory and definitions

Biodiversity reflects the hierarchy of increasing levels of organization and complexity in ecological systems; namely at the level of genes, individuals, populations, species, communities, ecosystems and biomes. It is communities of living organisms interacting with the abiotic environment that comprise, and characterize, ecosystems. Ecosystems are varied both in size and, arguably, complexity, and may be nested one within another.

Application of the ecosystem model (Tansley 1935; Odum 1969) implies comprehensive understanding of the interactions responsible for distinctive ecosystem types, but unfortunately this knowledge is rarely available. As a result, the use of the term ecosystem, when describing entities such as forests, grasslands, wetlands or deserts is more intuitive than based on any distinct spatial configuration of interactions.

Where communities of organisms persist in dynamic equilibrium over long periods of time and occupy the same physical space, ecosystems may appear to have discrete physical boundaries, but these boundaries are porous to organisms and materials. Boundaries are, of course, most noticeable when there are major differences in the abiotic environment (for example lakes versus grasslands) and certainly some terrestrial ecosystems still extend over very large areas of the planet, for example savannah and tropical rainforests. Nevertheless, species abundance and species composition within these ecosystems always varies temporally and spatially. The population dynamics of species create temporal heterogeneity, while gradients in abiotic variables lead to spatial heterogeneity (Whittaker 1975) often over orders of magnitude (Ettama and Wardle 2002).

Ecosystem processes (Table 1.a) result from the life-processes of multi-species assemblages of organisms and their interactions with the abiotic environment, as well as the abiotic environment itself. These processes ultimately generate services when they provide utilities to humans (see Table 1.b). Alterations in biodiversity can result in very noticeable changes in ecosystem functioning: for example individual genes may confer stress tolerance in crops and increased productivity in agricultural ecosystems, and invasive species may transform fundamental ecosystem processes such as the nitrogen cycle (see section 3). The dimensions of biodiversity and its relationships to human well-being have been extensively addressed by Levin (2000), including both the services that biodiversity supports and the evolutionary genesis of biodiversity together with the ecological processes underlying patterns and trends.

The relationship between biodiversity and ecosystem functioning cannot be revealed by ecological studies of communities that focus on the structure and behaviour of species and populations at a location. What is needed in addition are studies that address the flux of energy and matter through the ecosystem. The measures used may be different: for example, community studies may employ indices measuring aspects of biodiversity, whereas ecosystem studies utilize measures of standing crop, or flux of nutrients. Both are important in the evaluation of ecosystem services. Services directly linked to primary plant productivity, e.g. provisioning of food, are measured in biomass per unit area, or nutrient content per unit biomass, whereas cultural services may require a measure of complexity of biodiversity at a suitable scale, e.g. species richness in spatial units within the landscape (Srivastava and Vellend 2005). However, this is not to say that such measures are mutually exclusive. For example, the service of biological pest control is best estimated both by measures of biodiversity in terms of insect predator guilds, and their temporal relative abundance.

Table 1.a. **Some examples of biological and physical processes and interactions that comprise ecosystems functions important for ecosystem services.** (From Virginia and Wall, 2000)

Ecosystem function	Processes
Primary production:	Photosynthesis Plant nutrient uptake
Decomposition:	Microbial respiration Soil and sediment food web dynamics
Nitrogen cycling:	Nitrification Denitrification Nitrogen fixation
Hydrologic cycle:	Plant transpiration Root activity
Soil formation:	Mineral weathering Soil bioturbation Vegetation succession
Biological control:	Predator-prey interactions

Table 1.b: Examples of relationships between biodiversity and ecosystem services.

Component of biodiversity	Example of ecosystem service (see also section 3)	Sources
Genetic variability	Medicinal products	Chai et al. (1989)
Population sizes and biomass	Food from crops and animals	Kontoleon et al. (2008)
Species assemblages, communities and structures	Habitat provision and recreation	Rosenberg et al. (2000)
Interactions between organisms and their abiotic environment	Water purification	Hefting et al. (2003)
Interactions between and among individuals and species	Pollination and biological control	Messelink et al. (2008)

In any community of organisms, some groups make the principal contribution to a particular process, and so contribute to the overall functioning of the ecosystem of which they are a part. Thus, the critical functions of communities of soil organisms are decomposition and nutrient and elemental cycling, whereas plant communities contribute biomass production through photosynthesis. Communities in the soil are intimately interlinked (through root-microbe interrelations) with vegetation, and faunal communities depend not only on primary plant production *per se* but on the composition and physical structure of plant communities for habitat. This linkage between above-ground and below-ground parts of ecosystems is fundamental in all cases, as exemplified by provisioning ecosystem services in low-input agriculture by the role of legumes within cropping cycles.

Box 1 illustrates some of the linkages between different communities of organisms in relation to their major functions. These interactions contribute both to the regulation of biomass in an ecosystem and to the diversity of species assemblages within communities.

Box 1: **Biotic communities and their major functions**

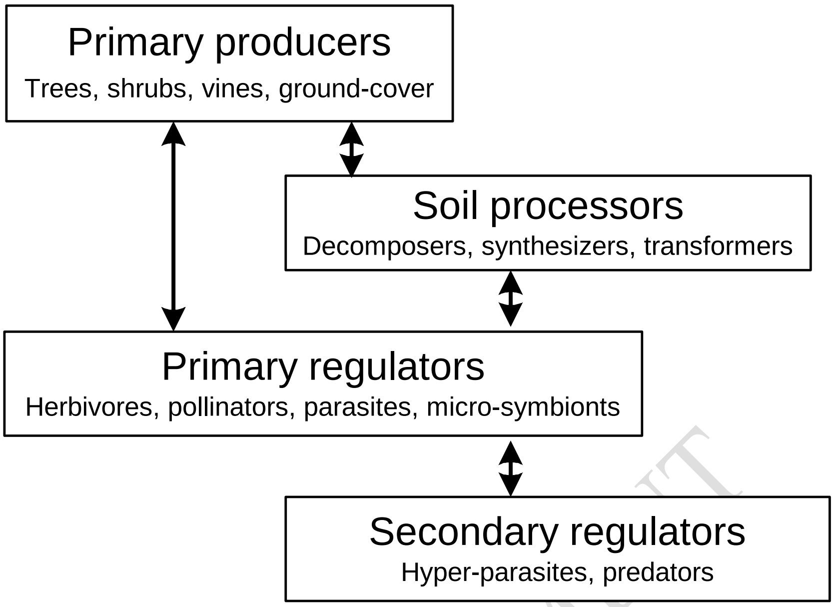

Figure 1: **Illustrative relationships between different functional groups in ecosystems.** (following Swift et al. 2004).

Primary producers:

Classification of plants into functional groups has an extensive history. Groupings can be based on a variety of reproductive, architectural and physiological criteria, but scale and efficiency of resource capture is often suggested as the main criterion. This will be determined by features of both architecture (e.g. position and shape of the canopy and depth and pattern of the rooting system) and physiological efficiency (see Smith et al. 1997). In some agro-ecosystems photosynthetic microorganisms may constitute a significant group, e.g. lowland rice.

Soil processors:

This is a very diverse community of organisms, involved in decomposition of organic matter (decomposers), soil synthesis (synthesizers) and nutrient cycling (transformers).

Decomposers:

This is a group of enormous diversity that can be subdivided taxonomically into bacteria, fungi, invertebrates, and others) having functional roles in the breakdown and mineralization of organic materials of plant or animal origin.

Synthesizers:

These are species that change the structure of soil and its porosity to water by burrowing, transport of soil particles amongst soil horizons, and formation of aggregate structures. Many of these species also contribute to decomposition.

Transformers:

This includes a range of autotrophic bacteria that utilize sources of energy other than organic matter (and therefore are not classifiable as decomposers) and play key roles in nutrient cycles as transformers of elements (carbon, nitrogen, phosphorus, sulphur etc.). Some heterotrophs that have a decomposer function but also carry out elemental transformations beyond mineralization (e.g. free-living di-nitrogen fixers).

Primary regulators:

Organisms that have a significant regulatory effect on primary production and therefore influence the goods and services provided by plants.

Pollinators:

Pollinators are a taxonomically very disparate group of organisms that includes many insect groups and vertebrates such as birds and bats.

Herbivores:

Vertebrate grazers and browsers are readily distinguished from invertebrate herbivores, although their impacts may be functionally similar and significant at the ecosystem level. The balance of effects of different types of herbivore can influence the structure of plant cover.

Parasites:

Microbial and fungal infections of plants may limit primary production in analogous manner to herbivory. Parasitic associations can also influence the growth pattern of plants and hence their architecture and physiological efficiency.

Micro-symbionts:

Mutualistic plant-microbial associations, e.g. di-nitrogen-fixing bacteria and mycorrhizal fungi.

Secondary regulators:

Hyper-parasites and predators:

This is diverse group of microbial parasites and vertebrate and invertebrate predators that feed on organisms in other groups and at other trophic levels.

Spatial interconnectedness maintains links and genetic interchange between populations of species, and underpins ecosystem functioning directly through physical connections. This is evident when considering energy and nutrient budgets; for example where nutrients ‘spiral’ downstream (Newbold et al. 1981) or move between floodplain wetlands and riverine ecosystems, especially due to flood ‘pulses’ (Junk et al. 1989). In this way, fish populations of African rivers benefit from the organic matter and nutrients deposited by both wild and domesticated herbivores grazing the floodplains during the dry season (Drijver and Marchand 1985). ‘Allochthonous’ organic matter (i.e. dead organic matter produced outside and transported into an ecosystem) may be important to the stability of ecosystems. At local scales dissolved or particulate organic matter may be dispersed by rivers during flooding (Junk et al. 1989). At larger scales, the annual migration of Pacific salmon (*Oncorhynchus* spp.) plays a key role in marine-freshwater nutrient recycling over vast distances (Mitchell and Lamberti 2005) with known dependencies for aquatic insect communities in Alaskan streams (Lessard and Merritt 2006), for brown bears *Ursus arctos* and for predatory birds (Hilderbrand et al 1999; Helfield and Naiman, 2006) and surrounding forest ecosystems. Polis et al. (1997) have highlighted the importance of understanding the impacts of nutrient transfers across ecosystem boundaries to the understanding of the dynamics of these systems.

The interactions within communities of organisms at population and community level play a key role in determining the stability and resilience of the ecosystem as a whole. Communities are structured by multiple biotic processes, and external conditions may strongly influence the outcome. Mouritsen et al. (1998) for example describe the dramatic impact of elevated summer temperatures on parasitic infections (by microphallid trematodes) on the mud snail *Hydrobia uvae* and amphipod *Corophium volutator* in Danish mudflats. High ambient temperatures in 1990 elevated the infection rate, which in turn led to the complete collapse of the amphipod population. The local extinction of this sediment-stabilizing population subsequently led to significant mudflat erosion and changes in topography. The result was substantive community depauperation, especially in macro-invertebrates, resulting in a change to the ecosystem (see also Griffin et al. 2009).

Understanding the role of biodiversity in ecosystem functioning has been considerably advanced by complementary studies of both the flow of energy and matter through trophic networks and the functional diversity of species within ecosystems (see Srivastava et al. 2009; Suding et al. 2008; Diaz et al. 2007a; Diaz and Cabido 2001). Villéger et al. (2008) have recently explored functional diversity indices that seek to encompass findings from both types of study. De Leo and Levin (1997) made a useful distinction between these two approaches. In practice, they are not mutually exclusive, and both underpin the ability of the ecosystem to support services of value to society. However, an increasing body of scientific evidence indicates that functional diversity, rather than species diversity *per se*, enhances ecosystem functions such as productivity (Tilman et al. 1997a; Hooper and Dukes 2004; Petchey et al. 2004), resilience to perturbations or invasion (Dukes 2001; Bellwood et al. 2004) and regulation of the flux of matter (Waldbusser et al. 2004).

Some species have a disproportionate influence on ecosystem functioning relative to their biomass and abundance, and the loss of such a 'keystone' species has cascading effects on community diversity and ecosystem functioning (Bond 1993). For example, the removal of the Pacific sea otter (*Enhydra lutris*) from Californian coastal ecosystems has led to the loss of the kelp community and many fish species; removal of fish-eating caiman from some areas of the Amazon resulted in a decline in the fish population and catch because of reduced nutrient cycling in the food chain (Williams and Dodd 1980); large changes in African elephant (*Loxodonta africana*) numbers have substantial effects on plant productivity, soil nutrient cycles and vegetation diversity in savannah woodlands and forests; and the impacts of herbivores on savannah plant communities are altered in ecosystems dominated by tsetse-flies.

A detailed discussion of functional traits, functional groups and functional diversity is provided by Hooper et al. (2005) where they concluded that:

I. Species functional characteristics strongly influence ecosystem properties. An increase in diversity leads to an increase in productivity due to complementary traits among species for resource use.

II. Increased biodiversity leads to an increase in response diversity (range of traits related to how species within the same functional group respond to environmental drivers) resulting in less variability in functioning over time (Elmqvist et al. 2003; Hughes et al. 2002).

III. Idiosyncratic effects due to keystone species properties and unique trait-combinations that may result in a disproportional effect of losing one particular species compared to the effect of losing one 'average' species.

1.2 The role of diversity in ecosystem functioning

In this section, we discuss issues of diversity and productivity and the roles of functional diversity before examining factors in ecosystem stability and change and the maintenance and generation of services.

1.2.1 Species diversity and productivity – terrestrial systems

Species dominating a community are generally major controllers of system function, yet evidence suggests that less obvious or abundant species have major roles in the functioning of ecosystems. These 'ecosystem engineers' (Swift et al. 2004), and 'keystone species' (Lyons *et al.*, 2005), may be uncommon species that greatly influence community dynamics, e.g. through enhancing resistance to species invasions (Lyons & Schwartz, 2001) or through their role as pollinators and seed dispersers (Cox et al. 1991). The population of an uncommon species may change dramatically in abundance and importance in response to particular conditions (Hobbs *et al.*, 2007), e.g in temperate lakes, species of plankton respond to seasonal changes in water temperature and mixing, and the associated availability of nutrients, resulting in rapid successional changes of species (Abrantes et al. 2006).

The diversity of functional types in soils is strongly linked to productivity. Many experiments have shown significant enhancements of plant production owing to the presence of soil animals, and specifically their diversity in the case of earthworms (Lavelle et al. 2006). The enhancement of primary production might be the result of increased release of nutrients from decomposition, enhancement of mutualistic micro-organisms (van der Heijden et al. 1998), protection against diseases, and effects on soil physical structure. However, experimentally removing key taxonomic groups from soil food webs may have little impact on rates of processes such as soil respiration and net ecosystem production (Ingham et al. 1985; Liiri et al. 2002; Wertz et al. 2006), possibly because the exceptional diversity of soil organisms and the relatively low degree of specialization in many groups means that many different species can perform similar processes (Bradford et al. 2002; Fitter et al. 2005).

The role of biodiversity in maintaining productivity has been studied in theoretical, controlled-environment and small- and large-scale field studies (see, for example, Naeem et al. 1995; Tilman et al. 1996, 1997b; Lawton et al. 1998), but few data are from “mature” natural ecosystems. Grace et al. (2007) compared a large set of natural ecosystems and suggested that the influence of diversity on productivity was weak when examined at small spatial scales. Nevertheless, a meta-analysis of published studies found clear evidence of a positive effect of biodiversity on productivity at the same trophic level where biodiversity was measured (Balvanera et al. 2006). Furthermore, Balvanera et al (2006) draw the following conclusions based on the review of current data: 1) plant diversity appears to enhance belowground plant and microbial biomass, 2) plant diversity has positive effects on decomposer activity and diversity, and both plant and mycorrhizal diversity increase nutrients stored in the plant compartment of the ecosystem, 3) increasing the diversity of primary producers contributes to a higher diversity of primary consumers, 4) higher plant diversity contributes to lowering plant damage by pest organisms, and 5) abundance, survival, fertility and diversity of invasive species is reduced when plant diversity increases. At large spatial scales, Costanza et al. (2007) showed that over half of the spatial variation in net productivity in North America could be explained by patterns of biodiversity if the effects of temperature and precipitation were taken into account.

In intensively managed and disturbed ecosystems, maximum productivity is typically achieved in systems of very low diversity, for example heavily fertilized monocultures. However, these systems require large inputs of resources, including fertilizers, biocides and water, which generally are not environmentally or economically sustainable (Wright 2008). Sustained high production without anthropogenic resource augmentation is normally associated with high levels of biodiversity in mature ecosystems. In an eight-year study, Bullock et al. (2007) reported positive effects of increased species richness on ecosystem productivity in restored grasslands on a range of soil types across southern England. Similarly, Potvin and Gotelli (2008) reported higher productivity in biologically diverse tree plantations in the tropics, suggesting that increasing diversity in timber plantations may be a viable strategy for both timber yields and biodiversity conservation.

1.2.2 Species diversity and productivity – marine systems

Biodiversity is also associated with enhanced productivity in marine systems (Worm et al. 2006). Arenas et al. (2009) examined how different components of biodiversity influence the performance of macroalgal assemblages in natural communities. They found positive relationships for biomass and species richness with productivity but also relationships of spatial aggregation and species evenness with some of the productivity-related variables analyzed. In a meta-analysis of published experimental data (Balvanera et al. 2006), it was found that increased biodiversity of both primary producers and consumers enhanced the ecosystem processes examined; the restoration of marine ecosystems has also been shown to increase productivity substantially. Overfishing together with climate change and other pressures are producing impacts of unprecedented intensity and frequency on marine ecosystems, causing changes in biodiversity, structure and organization of marine assemblages directly and indirectly (Worm et al. 2006). Numbers and diversity of large pelagic predators have been sharply reduced and the impacts of this loss can cascade through marine communities (Heithause et al. 2008). Predictions about how communities will respond to marine predator declines have to consider the risk effects and behaviorally mediated indirect interactions. In the case of vertebrate predators and long-lived prey species in particular, a sole focus on direct predation might greatly underestimate the community effects of predator loss (Heithause et al. 2008).

Although evidence from numerous experiments has very often shown a positive, but near universal saturating relationship between biodiversity and ecosystem functioning (Loreau 2008), analysis of deep sea ecosystems has shown a very different pattern. A recent global-scale study based on 270 datasets from 116 deep-sea sites, showed that functioning of these ecosystems is not only positively but also exponentially related to biodiversity in all the deep-sea regions investigated (Danovaro et al. 2008). Three independent indicators of ecosystem efficiency were used: 1) the meiofaunal biomass to organic C fluxes ratio, to estimate the system's ability to use the photic's zone primary production, 2) the prokaryote C production to organic C flux ratio, to estimate the system's ability to use and recycle organic matter deposited on the sea floor; and, 3) the total ratio of benthic meiofaunal biomass to sediment's biopolymeric C content, to estimate the system's ability to channel detritus to higher trophic levels. Significant and exponential relationships were found between biodiversity and each of these three independent indicators. Results suggest that higher biodiversity supports higher rates of ecosystem processes and an increased efficiency with which these processes are performed (Danovaro et al. 2008). These exponential relationships support the hypothesis that mutually positive functional interactions (ecological facilitation) are prevalent in these deep-sea ecosystems. Although there is still no full understanding of all the processes regulating deep-sea food webs and the ecological role of each species, it is hypothesized that the increase in bioturbation of the seafloor may increase benthic fluxes and the redistribution of food within the sediment, leading to an increase in ecosystem functioning. These results suggest that biodiversity loss in deep-sea ecosystems might be associated with significant reductions in functioning. Deep-sea sediments cover 65% of the world's surface, and deep-sea ecosystems play a key role in ecological and biogeochemical processes at a global scale. The importance of deep-sea biodiversity in maintaining the sustainable functioning of the worlds' oceans may still be grossly underestimated (Danovaro et al. 2008).

1.3 Functional groups and functional diversity

Functional groups are groups of organisms that perform particular operations in an ecosystem. They might, for example, produce biomass, pollinate, fix nitrogen, disperse seeds, consume other organisms, decompose biomass, mix soils, modify water flows, and facilitate reorganization and colonization. Loss of a major functional group may cause drastic alterations in ecosystem functioning (Chapin et al. 1997; Jackson et al. 2001). Hooper et al. (2005) concluded that certain combinations of species are complementary in their patterns of resource use and can increase average rates of productivity and nutrient retention, making diversity of functional traits one of the key controls on ecosystem properties.

Redundancy (i.e. more than one species performing the same process role) of functional traits and responses in ecosystems may act as an 'insurance' against disturbance and the loss of individual species if the diversity of species in the ecosystem encompasses a variety of functional response types (Hooper et al. 2005; Winfree and Kremen 2009). *Response diversity*, i.e. different responses to environmental change among species that contribute to the same ecosystem function, has been argued to be critical in ecosystem resilience (Elmqvist et al. 2003). Such species may replace each other over time, contributing to the maintenance of ecosystem function over a range of environmental conditions. Regional losses of such species increase the risk of large-scale catastrophic ecosystem shifts because spatial sources for ecosystem reorganization after disturbance are lost (O'Neill and Kahn 2000; Bellwood et al. 2004). This is a poorly understood area, but nonetheless current ecological theory predicts that when an ecosystem service is provided jointly by many species, it will be stabilized against disturbance by a variety of 'stabilizing mechanisms'. Few studies have investigated the occurrence of stabilizing mechanisms in landscapes affected by human disturbance. Winfree and Kremen (2009) used two datasets on crop pollination by wild native bees to assess three potential stabilizing mechanisms: density compensation (negative co-variance among species' abundances); response diversity (differential response to environmental variables among species); and cross-scale resilience (response to the same environmental variable at different scales by different species). They found evidence for response diversity and cross-scale resilience, but not for density compensation, concluding that these mechanisms may contribute to the stability of pollination services, thus emphasizing the insurance value of seemingly 'redundant' species.

1.4 The complexity of finding quantitative links between biodiversity and ecosystem services

In principle it should be straightforward to relate biodiversity measures to ecosystem service delivery, but in practice it is complicated by several factors (see also Chapter 3). First, biodiversity is a multidimensional concept and its description and measurement therefore takes many forms. Descriptions of biodiversity include classifications of the various hierarchical levels (communities, species, individuals, genes) but also of other dimensions such as interaction webs (trophic, host-parasite, pollinator), evolutionary diversity based on phylogenetic trees, trait diversity based on species-specific traits, or composite measures that attempt to summarize multiple measures. Some of these measures have been developed with a particular purpose in mind, others are attempts to simplify the complexity.

The second problem relates to the diverse set of purposes for the various measures of biodiversity that have been developed. Most available measures have been developed for specific purposes, so the available measures may not be what are needed for a particular purpose. For example, many available data sets that show large-scale (global, continental, major biome) distributions of biodiversity are measures of species richness, primarily derived for conservation reporting and planning, and tend to be counts of species richness or measures of population trends for large-bodied animals and plants. At smaller spatial and geographical scales, information is more varied, but again it is often information gathered for particular purposes (e.g. national reporting to international bodies for food and agricultural production and trade, conservation reporting, environmental quality monitoring). Therefore, most of the available data have been collected for another purpose, and are not obviously applicable to measures of biodiversity change that can inform analyses of ecosystem service delivery.

The third problem is that, although ecosystem service delivery often increases in quality, quantity or resilience with increasing biodiversity, the strength and the form of the relationship, and the measure of biodiversity that is the best predictor of ecosystem service quality or quantity, varies widely according to the ecosystem service being considered.

The above considerations mean that it is not yet possible to account accurately for the role of biodiversity, nor the probable impact of its decline, on ecosystem service delivery in general. On the one hand, measures of species richness (and subsets such as endemism, rarity, threat etc.), which are available globally for vertebrates and some plant groups, are hard to link directly to ecosystem functions and processes. On the other hand, locally available, ecosystem-specific or taxon-specific measures of functional type or functional diversity may relate well to certain specific ecosystem functions, but may not be generally applicable to other valued services in that ecosystem. Unfortunately, these local measures cannot be scaled-up to larger areas or transferred to other ecosystem types.

The extent to which biodiversity metrics can be used for ecosystem service assessments is therefore a direct consequence of whether the measures are correct for the context. Unfortunately, because the understanding of the role of biodiversity is still incomplete, one can only be confident about a few cases where good data are available that are known to support ecosystem service valuations. For example:

The productivity of terrestrial and aquatic systems for marketed foods, fuels or fibres can be measured using production statistics. The relevant measures of diversity in arable systems, for example, relate to crop genetic diversity, the diversity of land races and wild relatives, and the diversity of pests, pathogens, predators and symbionts. The most relevant biodiversity metric for crops is genetic diversity.

The ecosystem service of food production depends in many cases on pollinators. Here the relationship between the service and biodiversity is strong, and the relevant metric is pollinator species richness. While the form of these relationships may be quite general, it appears that the resistance of different areas to pollinator loss varies quite widely according to the nature of the plant-pollinator interaction web in that ecosystem, and the recent history of pollinator and plant decline.

Many cultural services depend primarily on species diversity, and tend to concentrate on the large-bodied, charismatic plants, birds and mammals. The relationships between the service and biodiversity in these cases are very strongly dominated by diversity measures that never saturate. In fact the values increase with the addition of more, rare forms. For these purposes, the global conservation species datasets are useful and highly relevant. However, the relationships do not scale down simply within countries or local areas.

The ecosystem service of freshwater quality shows a weak but rapidly saturating relationship with biodiversity and is strongly focused on a few functional types that are likely to be generally applicable across both scales and systems.

Some work done on ecosystem processes such as primary productivity or decomposition (referred to as supporting services in the Millennium Ecosystem Assessment (MA 2005)) may also be relevant for many ecosystem services that ultimately depend on them. In studies, plant functional traits such as leaf area or plant size are strong predictors of ecosystem process strength, and measures such as the weighted mean of the plants in the community are the best predictor, though sometimes the presence or absence of particular trait values are also very significant (Diaz et al. 2007b; Suding et al. 2008).

2 The links between biodiversity, ecosystem functions and ecosystem services

The following review of the evidence base for links between biodiversity, ecosystem functions and specific ecosystem services is based on two recent reviews, Balmford et al. (2008) and the EASAC report "Ecosystem services and biodiversity in Europe" (EASAC 2009) and updated with additional studies and reports. Substantial knowledge gaps remain, and understanding of the underlying processes for the generation of several services is limited; the following presentation reflects this variable knowledge. This section follows the general typology of services presented in chapter 1 and treats the services one by one, with the potential linkages among multiple ecosystem services further discussed in section 4. The typology where services are classified as *provisioning, regulating, habitat and cultural* is mainly used as a way of structuring information and does not reflect the inherent complexity where, e.g. a provisioning service, like fish, is not just representing a protein source, but also carries a strong cultural dimension related to harvesting techniques, preparation, symbolism etc. To place cultural values in a separate category is thus underestimating the cultural dimension of many of the services in other categories and this should be an area for further development.

PROVISIONING SERVICES

2.1 Provision of food

Context and importance of service

Agro-ecosystems provide food for human consumption and, together with the associated ecosystems supporting marine and freshwater fisheries, underpin global food security. Today 35% of the Earth's surface is used for growing crops or rearing livestock (MA 2005). Grazing land alone accounts for 26% of the Earth's surface, and animal feed crops account for a third of all cultivated land (FAO 1999). Heywood (1999) estimated that well over 6,000 species of plants are known to have been cultivated at some time or another, and many thousands that are grown locally are scarcely or only partly domesticated, whilst as many, if not more, are gathered from the wild. Despite this, only about 30 crop species provide 95% of humanity's food (Williams and Haq 2002) and it has been argued that the world is currently over-dependent on a few plant species.

Plants and animals derived directly from marine biodiversity provide a significant part of the human diet. Fisheries and aquaculture produced 110 million tonnes of food fish in 2006, a *per capita* supply of 16.7 kg (FAO 2009). Almost half of this (47 %) was produced by aquaculture. For nearly 3 billion people, fish represent at least 15% of their average *per capita* animal protein intake. Whereas official statistics estimate that in low-income food-deficit countries, the contribution of fish to the total animal protein intake was <20%, the true proportion is probably higher in view of the under-recorded contribution of small-scale and subsistence fisheries (FAO 2009).

Sensitivity of service to variation in biodiversity – terrestrial agro-ecosystems

Harlan (1975) argued that the increasing dependence on fewer species for crops was leading to the loss of native genetic resources, and higher yielding modern varieties were displacing 'landraces' uniquely adapted to local conditions. In genetic terms, landraces are typically heterozygous at many loci, and this *in-situ* gene pool, together with that in wild crop relatives, remains an essential source of genetic diversity for plant breeders for new varieties. Failure to maintain sufficient genetic diversity in crops can incur high economic and social costs. The potato famine in Ireland in the nineteenth century is generally attributed to the low genetic diversity of the potatoes cultivated there, making the entire crop susceptible to potato blight fungus, a problem resolved by using resistant varieties from original gene pools in South America. Mixtures of varieties may successfully reduce disease incidence and increase yields as for example with the case of barley in Europe (Hajjar et al. 2008; see general review in de Vallavieille-Pope 2004), although there is much variation and often conflicting conclusions are drawn.

Hooper and Chapin (2005) argue that maintenance of high productivity over time in monocultures almost invariably requires heavy and unsustainable subsidies of chemicals, energy, and financial capital (EASAC 2009). They suggest that, from both economic and ecological perspectives, diversity must become increasingly important as a management goal. Organic farming can increase biodiversity (species richness and abundance), but with inconsistent effects among organisms and landscapes (Bengtsson et al. 2005). Even though crop yields may be 20% lower in organic farming systems, inputs of fertilizer and energy may be reduced by 30–50%, and pesticide input by >90%, suggesting that the enhanced soil fertility and higher biodiversity found in organic plots may render these systems less dependent on external inputs (Mader et al. 2002). In addition, they may be as profitable, or more so, than conventional agro-industrial systems. However, reduced yields in organic farming results in a trade-off between land for agriculture and land for maintaining wild biodiversity. Biodiversity could be promoted by using intensive agriculture and devoting spare land to biodiversity or by extending 'organic' or integrated farming systems that promote biodiversity (Fischer et al. 2008), but the outcomes of these two approaches would be very different.

The value of biodiversity is evident in permanent grassland and pasture ecosystems, where increased species richness often enhances biomass productivity and ecosystem functioning (Bullock et al. 2007; Tilman et al. 1996, 1997a, b; Naeem et al. 1995). Such gains appear to exploit species complementarity (Cardinale et al. 2007), but may also reflect the 'sampling effect' (McNaughton 1993) i.e. the relative higher frequency of the more productive species in a mixture.

Sensitivity of service to variation in biodiversity – marine systems and aquaculture

With dwindling marine fish stocks worldwide, aquaculture is thought to be the way to increase fish production necessary to feed an increasing human population. But this activity, which has been growing rapidly and accounts now for half of the global fish production, is still very dependent on wild fish for seed and feed (FAO 2009) and thus on functioning natural ecosystems and biodiversity. Intensively cultured fish and shrimp are fed on fish meal and fish oil that comes mainly from fishing (Deutsch et al. 2007). Furthermore, most aquaculture uses other ecosystem services, especially nutrient recycling and water purification. Since they are concentrated in coastal areas, strong impacts are already being felt in some places (e.g. Chile, Thailand) and this has made the expansion of aquaculture difficult. Although much research has been devoted to the replacement of fish meal and fish oils with land plant-based materials (e.g. soy meal and other cereals), with very good results (Carter and Hauler 2000; Clayton et al. 2008), provision of these foodstuffs themselves has important environmental impacts (Fearnside 2001; Steinfeld et al. 2006; FOE 2008), and their diversion to fish food has nutritional costs for many poor people (Delgado et al. 2003) with high social costs. The use of seaweeds harvested from natural ecosystems or cultivated in seawater (e.g. Valente et al. 2006) may be a way to produce feed for herbivorous fish without burdening fisheries or agricultural land.

Where are services generated?

Food is produced principally in intensively managed agro-ecosystems, but apart from areas devoted to wildlife conservation or recreation, and those used for other production systems (e.g. forestry), most landscapes/seascapes are involved in food production to some extent. Urban and suburban areas have allotment and other forms of gardens that are used for food production, particularly in developing countries. The ubiquity of agricultural production also means that other ecosystems are frequently adjacent to food-producing land and processes and practices of agriculture may therefore have a broader impact. This may involve spray drift of pesticides, nutrient pollution and barriers to the migration and dispersal of organisms among remaining patches of non-agricultural land, with negative consequences for the ability of distributed populations to withstand environmental change.

Uncertainties in delivery of service

At current levels of consumption, global food production will need to increase by 50% within the next four decades to meet the demands of a growing human population (UN 2009) and as consumption levels and world food prices rise, pressure to maximize the area under production will grow. Given the rapidly growing demands on the planetary ecosystems (Rockström et al. 2009), it is becoming critical to understand how a dramatic increase in agricultural production and shifting land use in combination with climate change will affect natural processes of the biosphere and levels of key regulating ecosystem services (e.g., CO_2, nitrogen flow, freshwater consumption). Large uncertainties remain about the outcome of these complex interactions. Increasing offshore aquaculture for the production of fish and seaweeds for food will result in substantial intensification of the use of the sea for food production and since the open sea is usually poor in nutrients, these will have to be added (with deep-sea water or artificial fertilization). The effects of these practices for the open sea ecosystems and processes are poorly understood.

2.2 Water provision (2), including regulation of water flows (10) and water purification (11)

Context and importance of service – water provisioning

Ecosystems play important roles in the global hydrological cycle, contributing to water provision (quantity, defined as total water yield), regulation (timing, the seasonal distribution of flows) and purification (quality, including biological purity as well as sediment load) (Dudley and Stolton 2003; Bruijnzeel 2004; Brauman et al. 2007). Global water use is dominated by agricultural withdrawals (70% of all use and 85% of consumptive use), including livestock production, followed by industrial and domestic applications. Vegetation, particularly forests, significantly influences the quantity of water circulating in a watershed. It is commonly assumed that forests generate rainfall and, in comparison with pasture and agriculture, promote higher rates of evapotranspiration and greater aerodynamic roughness, leading to increased atmospheric humidity and moisture convergence, and thus to higher probabilities of cloud formation and rainfall generation. Although evidence is increasing (Bruijnzeel 2004) that large-scale land use conversions affect cloud formation and rainfall patterns, this effect is highly variable and specific. The hypothesis of a 'biotic pump' has been elaborated by Makarieva et al. (2006) and Makarieva and Gorshkov (2007) as an explanation of high rainfall in continental interiors of the Amazon and Congo river basins. Marengo et al. (2004) discussed the role of the Amazonian 'water pump' (see Chapter 1, Figure 7), assumed to sustain rain-fed agriculture and other ecological systems elsewhere in the continent. Shiel and Murdiyarso (2009) reviewed the mechanisms and proposed that if the 'water pump' hypothesis proves accurate, modest forest loss may transform conditions in continental interiors from moist to arid, and forest biodiversity may be an underestimated factor in regional rain fall regulation.

Context and importance – water regulation and purification

In areas with seasonal rainfall, the distribution of stream flow throughout the year is often of greater importance than total annual water yield. This is particularly important to agricultural production, as irrigation is most important during the dry season. The same conditions that increase water infiltration also result in lower surface run-off. The link between regulation of water supply and water quality is strong because rapid flows of water through soil or ecosystems reduce the time in which transformations can occur; extreme weather events thereby lead to poorer water quality.

Sensitivity of services to variation in biodiversity

Although vegetation is a major determinant of water flows and quality, and micro-organisms play an important role in the quality of groundwater, the relationship of water regulation and purification to biodiversity is poorly understood, except in so far as the states of soil and vegetation determine water flows and storage. The activity of soil organisms has a large and direct impact on soil structure and hence on infiltration and retention rates. Ecosystems such as forest and wetlands with intact groundcover and root systems are considered very effective at regulating water flow and improving water quality. Vegetation, microbes, and soils remove pollutants from overland flow and from groundwater through various means, including: physically trapping water and sediments; adhering to contaminants; reducing water speed to enhance infiltration; biochemical transformation of nutrients; absorbing water and nutrients from the root zone; stabilizing eroding banks; and diluting contaminated water (Brauman et al. 2007). Changes to water quality that occur in soil include the transformations of persistent organic pollutants (POPs), sequestration and conversion of inorganic ions (nitrate, phosphate, metals), and removal of disease-causing microbes such as *Cryptosporidium* (Lake et al. 2007). Similar processes, including nutrient uptake and consumption of pathogens, occur in water bodies, including lakes and rivers of good ecological quality.

Where are services generated?

Water reaches freshwater stores (lakes, rivers, aquifers) by a variety of routes, including direct precipitation, surface and subsurface flows, and human intervention. In all cases, the water quality is altered by the addition and removal of organisms and substances. Ecosystems therefore play a major role in determining water quality. In particular, the passage of water through soil has a profound impact, both through the dissolution of inorganic (for example nitrate, phosphate) and organic (dissolved organic carbon compounds, pesticides) compounds and the modification of many of these by soil organisms. This service is therefore relevant to all terrestrial ecosystems, but may be of particular significance in urban and intensively managed ecosystems.

Uncertainties in delivery of service

Most changes to the capacity of ecosystems to regulate and provide freshwater seem to derive from, and be generally proportional to, land-use change. However, in some situations a relatively small additional change may trigger a disproportionate – and sometimes difficult to reverse – response from ecosystems' hydrological function (Gordon et al. 2008). For example, human-induced eutrophication can lead to sudden shifts in water quality from clear to turbid conditions, due to algal blooms (Scheffer et al. 1993) which affect freshwater fisheries and recreational use of water bodies. Reduction of nutrient concentrations is usually insufficient to restore the original state, with restoration necessitating very substantially lower nutrient levels than those at which the regime shift occurred (see section 5.1 below). Another example is represented by cloud forest loss, which results in a regime shift that may be largely irreversible. In some areas, such forests were established under a wetter rainfall regime, thousands of years previously. Necessary moisture is supplied through condensation of water from clouds intercepted by the canopy. If the trees are cut, this water input stops and the resulting conditions can be too dry for recovery of the forest (cf. Folke et al. 2004). In addition, climate change potentially can trigger sudden changes, particularly in regions where ecosystems are already highly water-stressed.

2.3 Fuels and fibres

Context and importance of service

The provision of fuels and fibres – such as timber, cotton, jute, sisal, sugars and oils - has historically been a highly important ecosystem service. Natural systems provide a great diversity of materials for construction and fuel, notably oils and wood that are derived directly from wild or cultivated plant species. Production of wood and non-wood forest products is the primary commercial function of 34% of the world's forests, while more than half of all forests are used for such production in combination with other functions, such as soil and water protection, biodiversity conservation and recreation. Yet only 3.8% of global forest cover corresponds to forest plantations, indicating that a substantial fraction of natural forests is used for productive uses (FAO 2006).

There is currently intense interest and strong policy direction to increase the proportion of energy derived from renewable sources, of which biological materials are a major part. At present, this is being achieved partly by the cultivation of biomass crops and partly by diversion of materials otherwise useable as food for people or animals, including wheat and maize, to manufacture ethanol as a replacement for petrol and other oil-derived fuels. Recently a big effort has been put into the cultivation of algae for biofuels. Although most attempts at cultivation have selected microalgae known to have high oil content, some studies using macroalgal biomass are also underway (Ross et al. 2008; Adams et al. 2009). This production, which does not need arable land or freshwater, may be a way to produce clean energy without the social costs of terrestrial alternatives. The wider environmental impacts of these cultivations, however, will have to be determined, since they would be very large scale operations. In this context, Hill et al. (2006) have argued that biodiesel, in comparison to bioethanol, returns such significant environmental advantages that it deserves subsidy.

Sensitivity of service to variation in biodiversity

As in the case of food production, the mix of species cultivated in production forests is selected to maximize the rate of return on timber production, and does not generally reflect the range of ecosystem services that are co-produced with timber – watershed protection, habitat provision, climate amelioration and so on. Managed forests, like farms, typically depend upon a small number of species. The question of whether forests are more productive in terms of biomass if they have higher tree species diversity has been addressed by a few studies, with mixed results. For example, tree species diversity was found to have a negative relationship with above-ground biomass in natural forests of Central Europe (Szwagrzyk and Gazda 2007), no relationship with productivity in Aleppo pine and Pyrenean Scots forests of Spain (Vilà et al. 2003), and a positive effect on wood production in early successional Mediterranean type forests (Vilà et al. 2007). Although species diversity might lead to higher productivity in the forest, the proportion of commercial species in more diverse sites is typically lower (FAO 2006). On the other hand, species richness has been found to increase yields in tropical tree plantations, due to increased growth of individual trees (Potvin and Gotelli 2008), and it may reduce the impact of pests on timber species. At present, however, commercial timber production is dominated by a small number of species.

For biofuels, it seems unlikely that biodiversity of the crop will play a direct role in most production systems, although all land-based biofuel production will still rely on the supporting and regulating services, such as nutrient and water cycling, for which biodiversity of soil organisms is important. The exception is the proposal to use mown grassland as a second-generation biofuel. Sustained production in such a system may well be best achieved by a diverse mixture of plant species. Biofuel production with algae is dependent on aquatic biodiversity for the provision of species adapted to the different places where cultivations would be held.

Where are services generated?

Most ecosystems are important, including forests, savannas, grasslands and marine and coastal systems in delivering this service. Ecosystems likely to be used for biofuel production include forests, arable land generally and grasslands. There is likely to be strong pressure to bring land currently regarded as marginal for agriculture into production for biofuel production; because time-to-market issues are less important than for food production systems. Remote and relatively inaccessible areas where land values are low may be targets for biofuel systems, introducing conflicts with recreation and biodiversity conservation.

Uncertainties in delivery of service

It is likely that a decline in the provision of wild timber, plant fibres and fuelwood will take place in proportion to the decline in the forested area. Fragmentation, however, may result in a much quicker decline in forest productivity than what would be expected given the total area of remaining forest (Laurance et al. 2001). Climate change has also been implicated in increasing forest fire risk (e.g. Westerling et al. 2006) and the combined effects of fragmentation and climate change may conspire to prompt an abrupt increase in fire risk, which may be particularly devastating (and less likely to be reversible) in tropical rain forests, as species are not ecologically adapted to fire, and each fire event tends to increase the likelihood that future fires will take place.

2.4 Genetic resources

Context and importance of service

Genetic diversity of crops increases production and decreases susceptibility to pests and climate variation (Ewel 1986; Altieri 1990; Zhu et al. 2000). In low-input systems especially, locally adapted varieties often produce higher yields or are more resistant to pests than varieties bred for high performance under optimal conditions (Joshi et al. 2001). In agriculture, the diversity of genetic resources comprises the traditional resources (wild types and the older domesticated landraces) together with modern cultivars. Genetic resources will be increasingly important in support of improved breeding programs (e.g. for crop plants, farm animals, fisheries and aquaculture), with a wide range of objectives for increasing yield, resistance to disease, optimization of nutritional value, and adaptation to local environment and climate change. Advances in genomics research are opening up a new era in breeding, where the linkage of genes to traits (marker-assisted selection) provides a more efficient and predictable route than conventional breeding programs to improved strains.

Sensitivity of service to variation in biodiversity

This is a service for which biodiversity is of central importance, because genetic diversity is inevitably lost when biodiversity declines. The greatest focus on genetic diversity as a service is in the protection of gene pools for agriculture. The Food and Agriculture Organization of the United Nations (FAO) has done much significant work at the global level to support characterization of genetic resources in the food crop, livestock, fisheries/aquaculture and forestry sectors, but quantifiable data on trend analysis in genetic resources are very limited and have been collected only for relatively brief periods. There are now numerous initiatives to collect, conserve, study and manage genetic resources *in situ* (for example growing crops) and *ex situ* (for example seed and DNA banks) worldwide. New techniques using molecular markers are providing new precision in characterizing biodiversity (at the level of molecular systematics and taxonomy) and the genetic diversity within collections – a significant aid to developing management strategy to identify gaps and redundancy (Fears 2007). Box 2 highlights the fundamental importance (option/insurance value) of this reservoir of genetic diversity to crop improvement for stress tolerance.

Box 2: Biodiversity at the gene level

A success story

In low-lying agricultural regions of the world, in Bangladesh and India for example, farmers suffer annual crop losses because of flooding of up to 4 million tons of rice (*Oryza sativa*) – enough to feed 30 million people – and the costs across the vast rain-fed lowland areas of Asia, as a whole, amount to about a billion dollars. Flood tolerance originally observed in a traditional Indian variety FR13A, and subsequently located with molecular markers and transferred into modern cultivars by conventional plant breeding (Xu et al. 2006), is conferred by a particular gene, the Sub1A-1 gene at the polygenic 'Submergence-1' (Sub1) locus. This gene halts the elongation of rice stems as a response to flooding, ensuring carbohydrate conservation for further growth when flood waters recede, and enhances yield over susceptible varieties (see Figure 2). Phylogenetic analysis has shown that this particular gene is also present in wild relatives *O. rufipogon* and *O. nivara* that persist in the wetlands of south and South East Asia. These wetlands, such as the Plain of Reeds in southern Vietnam, not only provide ecosystem services in regulation of water flow and quality but also act as a habitat for the evolution of genetic variation amongst *Oryza* species.

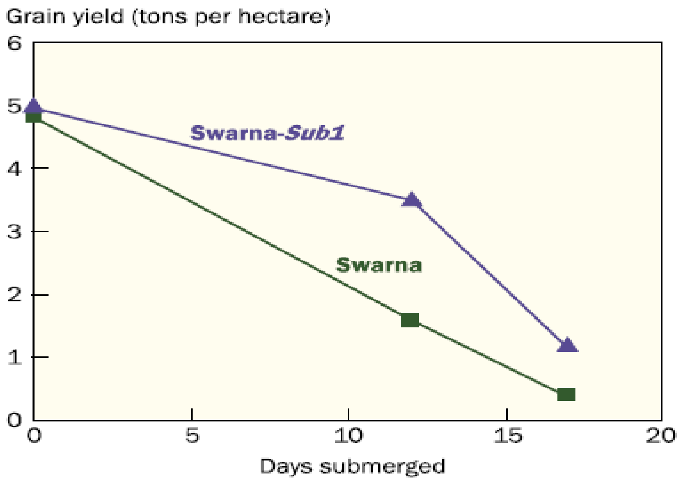

Figure 2: **The impact of insertion of the Sub-1 gene on the yield of the rice cultivar Swarna.** This gene confers tolerance to early submergence in water. Plants were completely submerged 14 days after the transplanting of 14 day-old seedlings in field trials at the International Rice Research Institute (Mackill 2006).

A current threat

The evolution of a new race (Ug99) of wheat stem rust (*Puccinia graminis*) in 1999 in the East African Highlands, and its subsequent range expansion from Kenya to Ethiopia has followed the predominant west-east airflows dispersing spores. It threatens global wheat production because of the absence of resistance in most modern cultivars. The potential migration path from East Africa via the Arabian peninsular, to the Middle East into the rice-wheat belt of the Indo-Gangetic plains represents a major threat to food security in South Asia. Strategies to mitigate the risks of loss of yield in a crop that underpins the livelihoods of millions of people requires the breeding of durable resistance into cultivars locally adapted for yield potential. Incorporating different combinations of race-specific resistance genes into new cultivars is one way forward. Such genetic diversity is present in germ plasm of wild relatives of wheat (e.g. *Triticum speltoides* and *T. monococcum*) and traditional Kenyan landraces (Singh et al. 2006).

Where are services generated?

All ecosystems are important for their genetic resources. Agricultural biodiversity can be considered to have a special status because of previous human efforts to improve varieties, hence the specific focus of the International Treaty on Plant Genetics Resources to conserve resources for food and agriculture. The replacement of landraces by high-yielding food crop varieties, taken together with other changes in agricultural practice has accelerated the erosion of genetic variation in cultivated material. The loss of genetic diversity associated with more intensive agriculture may also have deleterious impacts on the non-domesticated plants and animals (and micro-organisms) in the ecosystem. A decline in crop genetic diversity has consequences for their genetic vulnerability and their plasticity, for example to respond to biotic and abiotic stress.

Uncertainties in delivery of service

Given the likely non-linear relationship between area and genetic diversity, in some cases a small change in area (of natural habitat, or of traditional agricultural lands) may result in a disproportionate loss in genetic diversity of crops or livestock. This is probably more likely in areas that have already suffered extensive habitat loss and land conversion where the remaining populations of particular varieties and breeds are quite small. Climate change may also have non-linear effects on genetic diversity of crops and livestock.

2.5 Medicinal and other biochemical resources

Context and importance of service

Biochemicals encompass a broad range of chemicals of high value, for example metabolites, pharmaceuticals, nutrients, crop protection chemicals, cosmetics and other natural products for industrial use (for example enzymes, gums, essential oils, resins, dyes, waxes) and as a basis for biomimetics that may become increasingly important in nanotechnology applications as well as in wider contexts (Ninan 2009). Some of the best-characterized examples are pharmaceuticals, the value of which has been long recognized in indigenous knowledge. It has been estimated that "of the top 150 prescription drugs used in the U.S., 118 originate from natural sources: 74% from plants, 18% from fungi, 5% from bacteria, and 3% from one vertebrate (snake species)" (ESA 2000). In addition to these high-value biochemical products, there is an important related consideration in the use of biomass for chemical feedstocks in addition to bioenergy, where development of integrated biorefineries will generate the building blocks (platform chemicals) for industrial chemistry. A report from the US Environmental Protection Agency (2007) concludes that economically competitive products (compared with oil-derived) are within reach, for example for celluloses, proteins, polylactides, plant oil-based plastics and polyhydroxyalkanoates (Ahmann and Dorgan 2007). The high-value products may make use of biomass economically viable, leading to significant land-use conflicts.

Sensitivity of service to variation in biodiversity

Biodiversity is the fundamental resource for bioprospecting, but it is rarely possible to predict which species or ecosystem will become an important source. A wide variety of species – microbial, plant and animal – have been valuable sources of biochemicals, but the achievements so far are assumed to be only a very small proportion of what could be possible by more systematic screening. The impact of the current global decline in biodiversity on the discovery of novel biochemicals and applications is probably grossly underestimated. Biodiversity loss resulting from relatively low-value activities such as logging may compromise future high-value activities (as yet undiscovered) associated with the search for novel biochemicals and chemicals.

Where are services generated?

All ecosystems are potential sources of biochemicals. Numerous examples can be cited from the oceans and shoreline, freshwater systems, forests, grasslands and agricultural land. Species-rich environments such as tropical forests have often been assumed to supply the majority of products. However, the problem of the general lack of a robust and reliable measure to assess the commercial or other value of an ecosystem is compounded by the expectation that most biochemical resources have yet to be discovered and exploited. Microbes seem likely to be especially rich in undiscovered metabolic capacities, and the complexity of soil ecosystems indicates the potential in searching for novel biochemicals there.

Uncertainties in delivery of service

Species richness may be quickly reduced as habitat destruction progresses in highly diverse regions (e.g. Forest et al. 2007), and the sources of biochemicals may change abruptly e.g. in coral reefs going through a phase shift.

2.6 Ornamental resources

Context and importance of service

Biodiversity has played an iconic, ornamental role throughout the development of human society. Uses of plant and animal parts, especially plumage of birds, have been important in conferring individual status, position and influence. Ornamental plants are typically grown for the display of their flowers but other common ornamental features include leaves, scent, fruit, stem and bark. Considerable exploration effort, and some of the rationale of the voyages of discovery, was underpinned by the search for and transfer of species to be enjoyed in parks, gardens, private greenhouses and zoos by wealthy members of societies less endowed with biodiversity.

A modern example is provided by the statement by the Zoological Society of London that aquarium fish are the most popular pets in the world, representing an industry which in 1999 was worth $3 billion in annual retail sales. About 10% of the species are caught from the wild, causing concerns over the viability of stocks (ZSL 2006). Over 20 million freshwater fish are exported each year from the Brazilian Amazon and this generated $3 million in 2006 (Prang 2007). Birds are another focus of the ornamental value of biodiversity. In 1992, the trade in CITES (Convention on International Trade in Endangered Species)-listed wild birds was banned in the U.S., "leaving the EU responsible for 87% of the trade" (RSPB 2007). Because of fears for animal and human health, the EU issued a trade ban from July 2007, saving probably up to 2 million wild birds annually from the pet trade. Following the ban, the Royal Society for the Protection of Birds estimated that trade in CITES-listed threatened birds may drop from ca. 800,000 per year to a few hundred, because “import of small numbers of wild birds into the EU by zoos and some pet owners will still be allowed” (RSPB 2007).

Sensitivity of service to variation in biodiversity

The service is related completely to individual species and is highly sensitive to maintenance of viable populations.

Where are services generated?

The same applies as for service 3.5.

Uncertainties in delivery of service

The same applies as for service 3.5.

REGULATING SERVICES

2.7 Air quality regulation and other urban environmental quality regulation

Context and importance of service

Ecosystems contribute to several environmental regulation services of importance for human well-being, particularly in urban areas where vegetation may significantly reduce air pollution and noise, mitigate the ‘urban heat island’ effect (e.g. Santomouris 2001), and reduce impacts related to climate change (Bolund and Hunhammar 1999). This potential is often substantial (e.g. Pickett et al. 2008). For example, in the Chicago region, trees were found to remove some 5,500 tonnes of air pollutants per year, providing a substantial improvement in air quality (McPherson et al. 1997). Vegetation reduces noise levels, and dense shrubs (at least 5 m wide) can reduce noise levels by 2 dB(A), while a 50-m wide plantation can lower noise levels by 3–6 dB(A) (Bolund and Hunhammar 1999). Evergreen trees are preferred because they contribute to noise reduction year round (Ozer et al. 2008). Urban parks and vegetation reduce the urban heat island effect and have an important potential for lowering urban temperatures when the building envelope is covered with vegetation, such as green roofs and green walls, with the largest effect in a hot and dry climate (Alexandri and Jones 2007). In relation to overall climate change mitigation, urban ecosystems may assimilate non-negligable quantities of carbon, e.g. in Stockholm County, ecosystems assimilate about 17% of total anthropogenic CO_2 (Jansson and Nohrstedt 2001), and residential trees in the continental United States may sequester 20 to 40 teragrams C per year (Jenkins and Riemann 2003).

Green areas, vegetation and trees, also have direct health benefits, e.g. in a study from New York, presence of street trees was associated with a significantly lower prevalence of early childhood asthma (Lovasi et al. 2008). Green area accessibility has also been linked to reduced mortality (Mitchell and Popham 2008) and improved perception of general health (e.g. Maas et al 2006). In a review by Bird (2007), links were noted between access to green spaces and a large number of health indicators, e.g. coping with anxiety and stress, treatment for children with poor self-discipline, hyperactivity and Attention Deficit Hyperactivity Disorder (ADHD), benefiting elderly care and treatment for dementia, concentration ability in children and office workers, healthy cognitive development of children, strategies to reduce crime and aggression, strengthened communities, and increased sense of well-being and mental health. The distribution and accessibility of green space to different socio-economic groups, however, often reveals large inequities in cities (e.g. Pickett et al. 2008), contributing to inequity in health among socio-economic groups, although confounding effects are not always possible to separate (Bird 2007).

Sensitivity of service to variation in biodiversity

To what extent biodiversity and variation in species composition plays a role in the generation of environmental quality services is still poorly investigated (Elmqvist et al. 2008). For air quality, filtering capacity increases with leaf area, and is thus higher for trees than for bushes or grassland (Givoni 1991). Coniferous trees have a larger filtering capacity than trees with deciduous leaves (Givoni 1991). Figure 3 illustrates a hypothesized distribution of species richness in relation to degree of anthropogenic impact. The urban core has fewer species and often very different species involved in generation of ecosystem services than in more rural areas. Interestingly, the number of plant species in urban areas often correlates with human population size, and plant diversity may correlate positively with measures of economic wealth as shown for example, in Phoenix, USA (Kinzig et al. 2005).

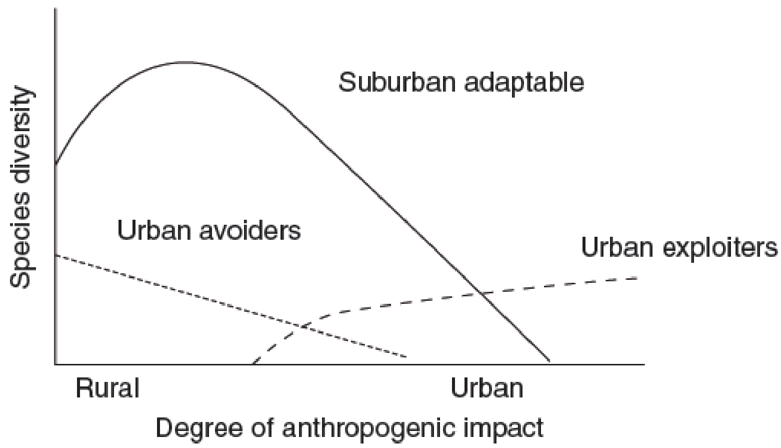

Figure 3: **Organisms may respond differently to increasing human impact.** Urban avoiders are large-bodied species or species linked to late successional stages. These species might be very sensitive and show a decline already at moderate human impacts. Suburban adaptable species may, to various degrees, utilize human modifications of the landscape; a large number of plant and animal species are likely to belong to this group. Urban exploiters directly benefit from human presence for food, reproduction or protection, and may often be cosmopolitan, generalist species. Source: Elmqvist et al. (2008).

Where are services generated?

Urban ecosystem services may be generated in a diverse set of habitats, including parks, cemeteries, vacant lots, streams, lakes, gardens and yards, campus areas, golf courses, bridges, airports and landfills. To what extent exotic species contribute to reduced or enhanced flow of ecosystem services is virtually unknown for any urban area, but since introduced species make up a large proportion of the urban biota, it is important to know not only to what extent introduced species are detrimental, but also to what degree some of the introduced species may enhance local diversity and maintain important functional roles.

Uncertainties in delivery of service

Considerable knowledge gaps remain about uncertainties and dynamics of urban ecosystem services. The Millennium Ecosystem Assessment (MA 2005), which covered almost every other ecosystem in the world, largely neglected urban systems, while on the other hand, the World Development Report (World Bank 2009), the world's largest assessment of urbanization, has left out ecosystems. Considerable uncertainties relate to the extent that isolation and fragmentation in the urban landscape influence the sustained generation of environmental quality services, and to the effects of climate change and rapid turnover of species on ecological functions of importance for these services.

2.8 Climate regulation

Context and importance of service

Climate is regulated on Earth by a natural 'greenhouse effect' that keeps the surface of the planet at a temperature conducive to the development and maintenance of life. Numerous factors interact in the regulation of climate, including the reflection of solar radiation by clouds, dust and aerosols in the atmosphere. In recent years the climate has been changing and the Earth is becoming warmer. Current change is driven largely by increases in the concentrations of trace gases in the atmosphere, principally as a result of changes in land use and rapidly rising rates of combustion of fossil fuels. The major greenhouse gas (CO_2) is absorbed directly by water and indirectly (through photosynthesis) by vegetation, leading to storage in biomass and in soils as organic matter; the ability of soils to store carbon is a major regulator of climate. Other greenhouse gases, notably methane (CH_4) and nitrous oxide (N_2O) are regulated by soil microbes. Organisms in the marine environment play a significant role in climate control through their regulation of carbon fluxes, by acting as a reserve or sink for CO_2 in living tissue and by facilitating burial of carbon in sea bed sediments (Beaumont et al. 2007). The capacity of the marine environment to act as gas and climate regulator is very dependent on its biodiversity.

An additional issue is the impact of vegetation on albedo: the reflection of incident radiation by land surfaces. Dark surfaces, especially those covered by evergreen forest, absorb more radiation than light surfaces, especially snow. Consequently, afforestation of boreal zones may lead to greater levels of warming, potentially outweighing the reduction expected from enhanced carbon sequestration by new trees.

Aerosols have a profound effect on climate, by intercepting and scattering radiation, and by acting as cloud condensation nuclei, thus reducing the amount of solar radiation reaching the Earth's surface. The production of aerosols by marine systems is well understood and has been taken into account in climate models. However, evidence is increasing that forests emit substantial amounts of biogenic volatile organic compounds, which can form aerosol particles. Forests are therefore simultaneously sinks for CO_2, sources of aerosol particles and determinants of albedo (Kulmala et al. 2004).

Sensitivity of service to variation in biodiversity

The interplay between biodiversity and climate regulation is poorly understood. The largest single store of carbon in terrestrial ecosystems globally is in the peat soils of the boreal and cool temperate zones of the northern hemisphere. The response of peatlands to climate change is crucial to predicting potential feedbacks on the global carbon cycle (Belyea and Malmer 2004). The climate-regulating function of peatlands also depends on land use because intensification of land use is likely to have profound impacts on soil carbon storage and on the emission of trace gases. Considering the area of drained and mined peatlands, restoration on abandoned mined peatlands may represent an important biotic offset through enhanced carbon sequestration (Waddington et al. 2001). However, mires are also major sources of potent greenhouse gases and the biodiversity of soil microbes is likely to play an important role in trace gas (methane) production (Roulet 2000). Current evidence for the role played by soil biodiversity in key processes leading either to carbon sequestration or to the release of trace gases is weak.

The exchange of CO_2 between atmosphere and ocean is larger than that between atmosphere and terrestrial ecosystems. Some of this occurs by physical processes, involving the equilibrium between CO_2 and carbonate, but a significant proportion is accounted for by biological processes. Although oceanic macrophytes account for less than 1% of global biomass carbon, the net primary productivity of the oceans is roughly equal to that of all terrestrial systems.

Where are services generated?

All soils store carbon, but to widely varying extents. The largest stores are in peatlands, but soils rich in organic matter occur in many ecosystems, especially where low temperature, low pH or waterlogging inhibit decomposition. Forests are the only major ecosystems where the amount of carbon stored in biomass of the plants exceeds that in the soil; deforestation therefore also affects climate regulation. Agricultural ecosystems currently have low soil carbon stores owing to intensive production methods, and there is scope for enhancing those stores. Marine ecosystems also play a major role in climate regulation through carbon sequestration and aerosol emission.

Uncertainties in delivery of service

Many uncertainties are associated with this service, particularly related to large time lags in the feedbacks between changes in ecosystem processes and the atmosphere. The global carbon cycle is strongly buffered, in that much of the CO_2 discharged by human activities into the atmosphere is absorbed by oceans and terrestrial ecosystems (Janzen 2004). However, the rate of emissions increasingly exceeds absorption capacity, and this capacity is at the same time reduced still further by anthropogenic damage to ecosystems. The complex interactions and long time lags make it very difficult to forecast eventual outcomes or if and when important thresholds will be passed.

2.9 Moderation of extreme events

Context and importance of service

Extreme events or 'natural hazards' are defined here as infrequent phenomena that may pose a high level of threat to life, health or property. Living organisms can form and create natural barriers or buffers, such as forests (including mangroves), coral reefs, seagrasses, kelp forests, wetlands, and dunes, and these can mitigate the effects of some natural hazards such as coastal storms (Wells et al.

2006), hurricanes (Costanza et al. 2006), catchment-borne floods (Bradshaw et al. 2007), tsunamis (Kathiresan and Rajendran 2005), avalanches (Gruber and Bartelt 2007), wild fires (Guenni et al. 2005) and landslides (Sidle et al. 2006). The available evidence for some of these effects is still scarce, and in some cases controversial. Many hazards arise from human interaction with the natural environment and are sensitive to environmental change. Examples include:

- flash floods due to extreme rainfall events on heavily managed ecosystems that cannot retain rainwater;
- landslides and avalanches;
- storm surges due to sea-level rise and the increasing use of hard coastal margins;
- air pollution due to intensive use of fossil fuels combined with extreme summer temperatures;
- fires caused by prolonged drought, with or without human intervention.

Sensitivity of service to variation in biodiversity

The role of biodiversity in delivering protection from natural hazards is generally small, but it has a role in facilitating recovery from such perturbations. In some particular cases, the ecological integrity of the affected ecosystem is of central importance, and it is likely that loss of biodiversity reduces resilience. Biodiversity plays a key role in the preservation of wetlands and coastal systems such as mangroves that deliver significant ecosystem services. For example, sea-level rise places intense selective pressures on halophytic vegetation whose fate is critical to the survival of salt marshes and other transition ecosystems (Marani et al. 2004). In mountain forests, increasing tree diversity is believed to enhance the protection value against, for example, rock fall (see, for example, Dorren et al. 2004).

Where are services generated?

Flooding is a problem in a wide range of ecosystems, including steep deforested catchments, flat alluvial plains and urban ecosystems with constrained water flows. Flooding can also occur because of exceptionally high tides and storm surges, a problem that will be exacerbated by rising sea levels; coastal wetlands are known to play a major part in defence against tidal flooding. Wind breaks from managed woods or from the use of natural forest features are a traditional means of protecting crops and habitations against both violent storms and general damage from exposure to high winds. In all these cases the role of vegetation is structural, and the part played by species composition will normally be indirect, in controlling the stability and resilience of the system.

Living marine flora and fauna can play a valuable role in the defence of coastal regions, and dampen and prevent the impact of tidal surges, storms and floods. This disturbance alleviation service is provided mainly by a diverse range of species which bind and stabilize sediments and create natural sea defences, for example salt marshes, mangrove forests, kelp forests and sea grass beds (Rönnbäck et al. 2007). Natural hazard regulation services show a declining trend due to loss of natural buffers such as wetlands and mangroves. For example, 20% or 3.6 million ha have been lost from the 18.8 million ha of mangrove forests covering the planet in 1980 (FAO 2007); 20% of coral reefs have been seriously degraded in only the past two decades (Wilkinson 2006); coastal wetland loss is extremely rapid, reaching 20% annually in some areas. On the other hand, the value of the regulation that is provided by these ecosystems is likely to be escalating, given an increase in human vulnerability to natural hazards.

Uncertainties in delivery of service

The effect of ecosystems on natural hazard mitigation is still poorly understood and it is uncertain to what extent abrupt changes in this service may be associated with abrupt changes in ecosystem extension and condition, for example the degradation of coral reefs or forests due to climate change. If the relationship between hazard regulation and ecosystem extension is an inverse asymptotic relationship, then regions where past ecosystem loss has been extensive may suffer a disproportionate future decline in the provision of this service.

2.10 Erosion prevention

Context and importance of service

Vegetation cover is the key factor preventing soil erosion, as classic historical examples such as the American dust bowl of the 1930s demonstrate, where lack of vegetation cover combined with drought resulted in unprecedented wind erosion, destroying farmland and livelihoods (Cooke et al. 1936). Landslide frequency seems to be increasing, and it has been suggested that land-use change, particularly deforestation, is one of the causes. In steep terrain, forests protect against landslides by modifying the soil moisture regime (Sidle et al. 2006).

Sensitivity of service to variation in biodiversity

This ecosystem service is generally not species specific or dependent on biodiversity in general, though in areas of high rainfall or extreme runoff events, forests may be more effective than grassland or herb-dominated communities.

Uncertainties in delivery of service

The same apply as for service 3.9.

2.11 Maintenance of soil quality

Context and importance of service

The process of soil formation is governed by the nature of the parent materials, biological processes, topography and climate. It involves the conversion of a mineral matrix, which has limited capacity to support nutrient cycles, into a complex medium with both inorganic and organic components, and solid, liquid and gas phases in which chemical and biological transformations take place. The progressive accumulation of organic materials is characteristic of the development of most soils, and depends on the activity of a wide range of microbes, plants and associated organisms (Brussaard et al. 1997; Lavelle and Spain 2001). Soil quality is underpinned by nutrient cycling, which occurs in all ecosystems and is strongly linked to productivity. A key element is nitrogen, which occurs in enormous quantities in the atmosphere and is converted to a biologically useable form (ammonium) by bacteria. Nitrogen fertilizer is increasingly expensive (about 90% of the cost is energy, typically from gas) and supplies are therefore not sustainable. Nitrogen fixation by organisms accounts for around half of all nitrogen fixation worldwide, and sustainable agricultural systems will have to rely on this process increasingly in future.

Sensitivity of service to variation in biodiversity

A large part of the organic material in many soils derives from the faeces of soil animals, and both the gross and fine structure of the soil is determined by biological activity. At a fine scale, structure may depend on fungal mycelia and the activities of mycorrhizal fungi, symbiotic with plant roots, which are the most abundant fungi in most soils (Miller and Jastrow 2000).

Many different species are implicated in nutrient cycling, which includes numerous transformations of elements often involving complex biochemistry. Nitrogen cycling may depend on diversity of plant communities and particularly on the presence of particular functional groups. Soil biodiversity has a particularly strong impact on nutrient cycling. Barrios (2007), in reviewing the importance of the soil biota for ecosystem services and land productivity, emphasized positive impacts of microbial symbionts on crop yields, as a result of increases in plant available nutrients, especially nitrogen, through biological nitrogen fixation by soil bacteria such as *Rhizobium*, and phosphorus through arbuscular mycorrhizal fungi.

Only 5–10% of added phosphate is recovered in crops, owing to its strong fixation by soils. In natural ecosystems, symbiotic mycorrhizal fungi are the main route of phosphorus transfer from soil to plant, and the diversity of mycorrhizal fungi can regulate both plant diversity and nutrient efficiency, and possibly water use efficiency (Brussaard et al. 2007). Sustainable agricultural systems will need to make greater use of mycorrhizal fungi, whose diversity is currently very low in arable systems (Helgason et al. 1998). It seems that functional diversity (and its influence on trophic interactions) rather than species diversity, is key to the decomposition, nutrient cycling, and stability of soil processes.

Where are services generated?

Soil formation is a continuous process in all terrestrial ecosystems, but is particularly important and active in the early stages after land surfaces are exposed (e.g. after glaciation).

Uncertainties in delivery of service

Agricultural expansion into new areas often occupies terrains that are not particularly suitable for agriculture, and soil fertility may decline very quickly as crops effectively mine the soil nutrients (Carr et al. 2006).

2.12 Pollination services

Context and importance of service

In some estimates, over 75% of the world's crop plants, as well as many plants that are source species for pharmaceuticals, rely on pollination by animal vectors (Nabhan and Buchman 1997). While the extent to which staple food crops depend on pollinator services has been questioned (e.g. Ghazoul 2005), Klein et al. (2007) found that, for 87 out of 115 leading global crops (representing up to 35% of the global food supply), fruit or seed numbers or quality were increased through animal pollination. In many agricultural systems, pollination is actively managed through the establishment of populations of domesticated pollinators, particularly the honeybee (*Apis mellifera*). However, the importance of wild pollinators for agricultural production is being increasingly recognized (e.g. Westerkamp and Gottsberger 2000; Kremen et al. 2007) and wild pollinators may also interact synergistically with managed bees to increase crop yields (Greenleaf and Kremen 2006).

Sensitivity of service to variation in biodiversity

Bees are the dominant taxon providing crop pollination services, but birds, bats, moths, flies and other insects can also be important. Studies in agricultural landscapes commonly show that increasing distances from forest fragments result in a decrease in both abundance and species-richness of flower-visiting bees (e.g. Steffan-Dewenter and Tscharntke 1999) and a recent quantitative review of 23 studies (Ricketts et al. 2008) found an exponential decay in pollinator richness and native pollinator visitation rate with distance to natural or semi-natural habitats. Hajjar et al. (2008) argue that the loss of biodiversity in agro-ecosystems through agricultural intensification and habitat loss negatively affects the maintenance of pollination systems, and causes the loss of pollinators worldwide (Kearns et al. 1998; Kremen and Ricketts 2000; Kremen et al. 2004; Richards 2001). Richards (2001) reviewed well-documented cases where low fruit or seed set by crop species, and the resulting reduction in crop yields, has been attributed to the impoverishment of pollinator diversity. Increasing evidence indicates that conserving wild pollinators in habitats adjacent to agriculture improves both the level and the stability of pollination services, leading to increased yields and income (Klein et al. 2003). Furthermore, a diverse assemblage of native pollinators provides insurance against year-to-year population variability or loss of specific pollinator species (Ricketts 2004; Tscharntke et al. 2005; Hoehn et al. 2008), and might better serve flowers because of pollinator-specific spatial preferences to a flowering plant or crop field (Klein et al. 2007). Given current declines in populations of managed honeybees (Johnson 2007), and abandonment of beekeeping in regions affected by 'Africanization' of honeybees (Brosi et al. 2007), the importance of wild pollination is likely to increase.

Where are services generated?

This service is important in all ecosystems, though possibly least important in species-poor boreal and arctic systems, where most species are wind-pollinated. Pollinator species often depend on natural or semi-natural habitats for the provisioning of nesting (e.g. tree cavities, suitable soil substrates) and floral resources that cannot be found within crop fields (Kremen et al. 2004). Consequently, the available area of natural habitat has a significant influence on pollinator species richness (Steffan-Dewenter 2003), abundance (Heard et al. 2007; Morandin et al. 2007), and pollinator community composition (Steffan-Dewenter et al. 2002; Brosi et al. 2007). Loss of suitable habitat is a key driver of declines in pollination services by wild pollinators, and habitat degradation through agricultural intensification leads to scarcity in critical floral and nesting resources for many species. In southern China, large areas of fruit orchards now need to be pollinated by hand since wild pollinators have disappeared, with approximately 10 people needed to do the work previously done by one bee colony.

Uncertainties in delivery of service

It is possible that a threshold in pollinator species functional diversity exists, below which pollination services become too scarce or too unstable to persist (Klein et al. 2007). Such a tipping point might occur when, at a landscape context, sufficient habitat is destroyed that the next marginal change causes a population crash in multiple pollinators. Alternatively, a threshold in habitat loss may lead to the collapse of particularly important pollinators, leading to a broader collapse in pollination services. Supporting this prediction, Larsen et al. (2005) found that large-bodied pollinators tended to be both most extinction-prone and most functionally efficient, contributing to rapid functional loss with habitat loss. Increased uncertainty is also represented by climate change, as phenological shifts may result in asynchrony and disruption of plant-pollinator interactions (Memmott et al. 2007).

2.13 Biological control

Context and importance of service

Pests and diseases are regulated in ecosystems through the actions of predators and parasites as well as by the defence mechanisms of their prey. Natural control of plant pests is provided by generalist and specialist predators and parasitoids, including birds, bats, spiders, beetles, mantises, flies, and wasps, as well as entomopathogenic fungi (Way and Heong 1994; Naylor and Ehrlich 1997; Zhang et al. 2007). In the short-term, this process suppresses pest damage and improves yields, while in the long-term it maintains an ecological equilibrium that prevents herbivorous insects from reaching pest status (Zhang et al. 2007, Heong et al. 2007). Agricultural pests cause significant economic losses worldwide. Globally, more than 40% of food production is being lost to insect pests, plant pathogens, and weeds, despite the application of more than 3 billion kilograms of pesticides to crops, plus other means of control (Pimentel 2008). The services of regulation are expected to be more in demand in future as climate change brings new pests and increases susceptibility of species to parasites and predators.

Sensitivity of service to variation in biodiversity

The diversity of natural enemies seems to improve biological control through a variety of mechanisms, including: i) species complementarity, when more than one type of predator or parasitoid adds to the control of a pest species; ii) the sampling effect, whereby a particularly effective natural enemy is more likely by chance alone to occur when more species are present; iii) redundancy, where more species will buffer against disturbance or ecosystem change; and iv) idiosyncrasy, when the whole is greater than the sum of the parts owing to interactions among species (Tscharntke et al. 2005; Kremen and Chaplin-Kramer 2007).

A diverse soil community will not only help prevent losses due to soil-borne pests and diseases but also promote other key biological functions of the soil (Wall and Virginia 2000). Soil-borne pests and diseases such as root-rot fungi cause enormous annual crop losses globally (Haas and Défago 2005), but bacteria in the rhizosphere (the soil surrounding roots) can protect plant roots from diseases caused by root-rot fungi (Haas and Keel 2003); similarly, symbiotic mycorrhizal fungi can protect roots from pathogenic fungi (Newsham et al. 1995). Plant-parasitic nematodes represent a major problem in agricultural soils because they reduce the yield and quality of many crops and thus cause great economic losses. However, nematodes have a variety of microbial antagonists that include nematophagous and endophytic fungi, actinomycetes and bacteria (Dong and Zhang 2006).

Where are services generated?

The natural control of diseases and invasions occurs in all ecosystems. Those heavily influenced by human activity incur the greatest risk of both disease outbreaks and invasion. Data on populations of biological control agents are scarce but the trends are presumed to be negative owing to habitat transformation associated with agricultural intensification (agricultural expansion, enlargement of field size, and removal of non-crop habitat, which results in a loss of the natural landscape features required for maintaining their populations) and increasing pesticide use. On the other hand, the increase in organic farming worldwide may help to reverse this trend (Bengtsson et al. 2005; Willer et al. 2008).

Uncertainties in delivery of service

The relationship between densities of natural enemies and the biological control services they provide is unlikely to be linear (Losey and Vaughan 2006) and biological control functions may decline disproportionately when a tipping point in natural enemy diversity is passed. Empirical evidence in support of this logic is scarce, but the importance of natural enemy assemblage composition in some instances of biological control (Shennan 2008) indicates that changes in composition can lead to disproportionately large, irreversible and often negative shifts in ecosystem services (Díaz et al. 2006).

HABITAT SERVICES

The habitat 'service' was identified in chapter 1 as a distinct category to highlight a) the interconnectedness of ecosystems in the sense that different ecosystems provide unique and crucial habitats for particular life-cycle stages of migratory species; and b) that certain ecosystems have been identified that are of unique importance in that they have been found to exhibit particularly high levels of genetic diversity of major importance to maintain life (genetic diversity) on Earth, and natural adaptation processes. Both of these features underpin all, or most, provisioning, regulating and cultural services (which is why they are often called "supporting services") but they are distinct services in their own right, as explained below, and depend on particular spatial conditions within ecosystems.

2.14 Maintenance of life cycles of migratory species

Context and importance of service

The life cycle of any species is supported in its entirety or in part by the products and behaviour of many others as well as by the nature of the abiotic environment. Products of ecosystems (e.g. nutrients, seeds) may be exported by wind, water or animals (including humans) to support life cycles of species elsewhere. These interactions between ecosystems should be taken into account when assessing the ecological or economic importance of a given area.

Migratory species, e.g. some species of fish, birds, mammals and insects, might use an ecosystem for just part of their life cycle. For example, salmonid fish use clean, aerated, shallow areas of flowing water for courtship and egg-laying, and are dependent on these ecosystems to supply clean water and food for juvenile fish (e.g. Kunz 2004). The adult salmon supports other predatory species, including humans and, on death, contribute significant quantities of organic matter to the river ecosystem. Migrating birds such as geese rely on ecosystems for availability of grazing on their migration 'flyways', and can shape vegetation community composition, affecting competitive interactions and potentially increasing spatial heterogeneity by means of selective feeding (van den Wyngaert and Bobbink 2009). Some of the migratory species have commercial value, in which case the ecosystem providing the reproduction habitat provides an important so-called 'nursery-service' which is (economically) valued in its own right (e.g. mangroves, providing reproduction habitat to many species of fish and crustaceans, which are harvested as adults far away from their spawning areas). When economically valuing mangrove-ecosystems, this nursery service should be taken into account.

Sensitivity of service to variation in biodiversity

A high level of interdependency exists among species, and any species loss has consequences, some of which remain unnoticed by human observers, while some will be significant for functioning and provision of ecosystem services for migrating species. Loss of biodiversity will inevitably result in loss of functioning, and consequently, loss or degradation of these ecosystem services (Naeem et al. 1995).

2.15 Maintenance of genetic diversity

Context and importance of service

Genetic diversity, both within and between species populations, is characteristic of all ecosystems and, through natural selection, results in evolution and adaptive radiation of species to particular habitats. The degree of genetic diversity present within a species (which can be expressed in a variety of ways (Nei 1987) will depend on both individual species breeding behaviour, the extent to which gene flow occurs between populations, and the biotic and abiotic forces driving selection, in addition to mutation events. Micro-evolution (for example of metals tolerance in grasses) can occur over remarkably short distances of a few metres and within a few generations (Antonovics and Bradshaw 1970). On the other hand, certain species are endemic to particular ecosystems and regions of the world (Morrone 1994), reflecting macro-evolution. Ecosystems that exhibit particularly high levels of biodiversity (biodiversity hotspots) with exceptional concentrations of endemic species, are undergoing dramatic habitat loss. "As many as 44% of all species of vascular plants and 35% of all species in four vertebrate groups are confined to 25 hotspots comprising only 1.4% of the land surface of the Earth" (Myers et al. 2000). In addition to the overall importance of these 'hotspots' in maintaining genetic diversity, this service is of particular and immediate importance in preserving the gene-pool of most of our commercial crops and livestock species. Gene banks, which represent a mechanism of conservation, do not include the processes that generate new genetic diversity and adaptations to environmental change through natural selection.

Sensitivity of service to variation in biodiversity

Preservation of this (remaining) genetic diversity in these hotspots is of strategic value to the provision of ecosystem services, since the hotspots themselves contain not only species richness and genetic diversity within species, but represent the natural laboratory in which evolution can occur. The relationships between biodiversity hotspots, endemism and extinction threat, however, remain a continuing debate in conservation biology (see Prendergast et al. 1993; Orme et al. 2005). The debate about *in-situ* versus *ex-situ* conservation of genetic resources has equivalent prominence in the preservation of sources of both crop and animal germplasm for breeding purposes (see earlier sections). Vavilov (1992) originally promoted the concept of centres of origin of cultivated plants, which recognized that particular temperate and tropical ecosystems were the source of genetic diversity from which crop domestication occurred. The loss of the genetic diversity within habitats within these ecosystems can only be partially balanced by *ex-situ* conservation in gene banks (Nevo 1998), which by their very nature prohibit the continued evolution among wild, feral and domesticated species in the field.

CULTURAL AND AMENITY SERVICES

2.16 3.18-22 Cultural services: aesthetic information, opportunities for recreation and tourism, inspiration for culture, art and design, spiritual experience, information for cognitive development

Context and importance of service

Cultural and amenity services refer to the aesthetic, spiritual, psychological, and other benefits that humans obtain from contact with ecosystems. Such contact need not be direct, as illustrated by the popularity of the virtual experience of distant ecosystems through books, art, cinema, and television. Nor need such contact be of a wild or exotic nature, as shown by the ubiquity of e.g. urban gardens (Butler and Oluoch-Kosura 2006). The classification here largely follows the one in the Millennium Ecosystem Assessment (MA 2005) although considerable debate remains about how the wide range of benefits derived from these services should be classified. It has been proposed that many of these services should more appropriately be placed under provisional services, being of similar importance as food, water, etc. for human well-being (K.Tidball pers. comm.). For convenience, these services are here considered as falling into two main groups: i) spiritual, religious, aesthetic, inspirational and sense of place; and ii) recreation, ecotourism, cultural heritage and educational services.

An economic value is hard to apply to those in the first group, while the second group is more amenable to traditional valuation approaches. Although all societies value the spiritual and aesthetic ‘services’ that ecosystems provide, these may have different significance in affluent, stable and democratic societies. Nevertheless, biodiversity plays an important role in fostering a sense of place in most societies and has considerable intrinsic cultural value. Although recent high-profile research suggests that nature recreation is declining *per capita* in US and Japan (Pergams and Zaradic 2008), this trend is not mirrored in much of the rest of the world, where growth in visitation to protected areas is growing at least as fast as international tourism as a whole. Fewer data are available for other types of outdoor activity, though it has been estimated that each year over half the population of the

UK makes over 2.5 billion visits to urban green spaces (Woolley and Rose 2004), and 87 million Americans participated in wildlife-related recreation in 2006, an increase of 13% over the decade (USFWS 2007). Wildlife-based marine tourism, as whale and dolphin watching, is also a profitable activity that is highly dependent on a functioning ecosystem (Wilson and Tisdell 2003).

Many cultural services are associated with urban areas, and strong evidence demonstrates that biodiversity in urban areas plays a positive role in enhancing human well-being (see section 3. 7). For example, Fuller et al. (2007) have shown that the psychological benefits of green space increase with biodiversity, whereas a green view from a window increases job satisfaction and reduces job stress (Lee et al. 2009). This may have a strongly positive effect on economic productivity and hence regional prosperity. Several studies have shown an increased value of properties (as measured by hedonic pricing) with proximity to green areas (Tyrväinen 1997; Cho et al. 2008). Nihan (2009) and Shu Yang et al (2004) have also pointed to the role of ecosystems in providing design features that can be utilized in the context of eco-design in architecture and urban and community planning

Sensitivity of service to variation in biodiversity

The role of biodiversity varies greatly among these services but is likely to be particularly large for ecotourism and educational uses of ecosystems. However, in many cases biodiversity may not be the typical identifier of the value being placed on the ecosystem, but nevertheless underlies the character recognized by the visitor.

Where are services generated?

Cultural and recreational services based on biodiversity are most strongly associated with less intensively managed areas, where semi-natural biotopes dominate, although in urban areas this may vary. Low-input agricultural systems are also likely to support cultural services, with many local traditions based on the management of land and its associated biological resources. Newly created or restored green spaces are becoming an increasingly important component of the urban environment providing this service.

Uncertainties in delivery of service

Uncertainty may be assessed for tourism, where abrupt changes in the provision of tourism benefits can occur for a range of reasons. Some of these may be ecological, as systems reach tipping points. Key wildlife populations may collapse through disease or other factors, fire may destroy picturesque landscapes, corals may bleach with sudden temperature shifts, ecosystems may suddenly change from one (attractive) to another (less desirable) stable state. Some of these will be reversible, others will be permanent. Abrupt shifts may also (and perhaps more often) be socially instigated. War, terrorism, socio-political disruption, natural disasters and health crises all tend to rapidly and negatively affect international tourism demand. Likewise, events such as the foot and mouth disease outbreak in the UK in 2001 had dramatic impacts as people were prevented from visiting the countryside for recreation. The current volability in oil prices (and thus aviation fuel costs) and potential carbon taxes may have similar impacts on international tourism if such changes are too sudden and result in an increase in recreational visits in areas closer to urban centres (for urban recreational services see section 3.7).

3 Managing multiple ecosystem services

3.1 Bundles of ecosystem services

Functioning ecosystems produce multiple services and these interact in complex ways, different services being interlinked or 'bundled' together, and therefore affected negatively or positively as one service (e.g. food) increases (e.g. Bennet et al. 2009). Most studies so far have focused on one or a few services such as pollination, or food versus water quality and quantity. Characterizing multiple ecosystem services as well as biodiversity across the same region has only recently emerged as a field of study (e.g. Schroter et al. 2005), and the little quantitative evidence available to date has led to mixed conclusions (e.g. Bohensky et al. 2006). Scientists have tended to use land use ⁄ land cover as a proxy for the provision of services (Nelson et al. 2009) even though the relationships between land use, land cover and service provision are largely untested for most services in most regions of the world (Naidoo et al. 2008). Finding ways of assessing how multiple ecosystem services are interconnected and coupled to each other in 'bundles' is one of the major research gaps on ecosystem services identified by the MA (Carpenter et al. 2009). Furthermore, finding ways to target and implement payments for biodiversity conservation with 'bundles' of ecosystem services, e.g. carbon and water services, also is a major priority (Wendland et al. 2009) that will be discussed in section 5.2 below.

3.2 Trade-offs

Some ecosystem services co-vary positively (more of one means more of another) e.g. maintaining soil quality may promote nutrient cycling and primary production, enhance carbon storage and hence climate regulation, help regulate water flows and water quality and improve most provisioning services, notably food, fibre and other chemicals. Other services co-vary negatively (more of one means less of another) such as when increasing provisioning services may reduce many regulating services e.g. provision of agricultural crops may reduce carbon storage in soil, water regulation, cultural services, etc.

Provisioning and regulating ecosystem services can have a range of possible trade-offs. Depending on the type of trade-off (A, B or C in Figure 4), the supply of regulating services can be low, intermediate or high for similar levels of provisioning services. This will have very different implications for the design and management of landscapes. For example, it has been suggested that major ecosystem degradation tends to occur as simultaneous failures in multiple ecosystem services (Carpenter et al. 2006). The dry lands of sub-Saharan Africa provide one of the clearest examples of these multiple failures, causing a combination of failing crops and grazing, declining quality and quantity of fresh water, and loss of tree cover. However, a synthesis of over 250 cases of investments in organic agriculture in developing countries around the world (both dry lands and non-dry lands) showed that the implementation of various novel agricultural techniques and practices could result in a reduction of ecosystem service trade-offs, and increased levels of regulating services, even as crop yields were maintained (Pretty et al. 2005) (corresponding to B or even C in Figure 4).

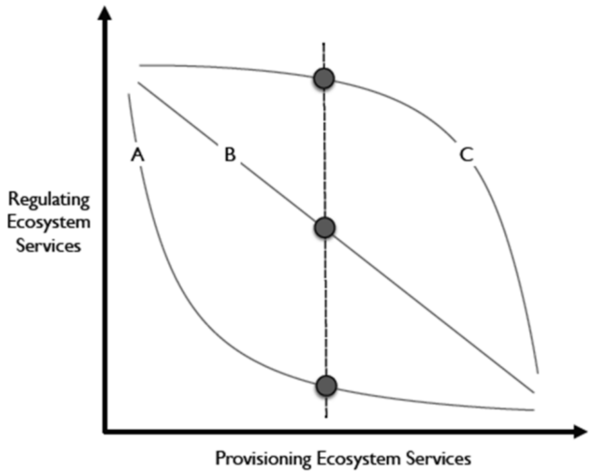

Figure 4: **Potential trade-offs between provisioning services and regulating ecosystem services.** A) Shifting an ecosystem to an increase in provisioning services produces a rapid loss of regulating services, B) regulating services linearly decrease with increases in provisioning services, and C) provisioning services can increase to quite high levels before regulating services decline. Source: Elmqvist et al. (2010).

The generation of some services may also result in other less desired effects, sometimes referred to as disservices, for example, the foods, fuels and fibres grown to satisfy basic human needs for nutrition and shelter may be highly valued, but the pests and pathogens deriving from the same ecosystems have a negative value. Both are products of the way in which the underlying ecosystems are managed, with the result that trade-off decisions have to be made (see Box 3). Knowledge of these relationships is essential to ensure that policy decisions translate into operationally effective and predictable outcomes.

Box 3: Trade-offs among ecosystem services

Several different types of trade-off can be identified, and are not mutually exclusive:

1. Temporal trade-offs: benefits now – costs later

Temporal trade-offs represent the central tenet of sustainable development "... that meets the needs of the present generation without compromising the needs of future generations….." (WCED 1987).

2. Spatial trade-offs: benefits here – costs there

Spatial trade-offs are behind much deliberation between communities and countries (especially water) and also occur between ecosystems and production landscapes. An example of a landscape level trade off is between improved water productivity (evapotranspiration used per tonne of grain) up stream and consequential down stream problems with deteriorating water quality associated with the use of agricultural inputs.

3. Beneficiary trade-offs: some win – others lose

These trade-offs are real but it is possible to move towards "winning more and losing less" by improving access to information on ecosystem services and their valuation, framing and using appropriate incentives and/or markets, and clarifying and strengthening rights of local people over their resources.

4. Service trade-offs: manage for one service – lose another

Manipulation of an ecosystem to maximize one particular service risks reducing others, e.g. maintaining monocultures of a single species (for production of food, fibre and energy), will reduce the delivery of services dependent on the maintenance of biodiversity, including pollination and disease regulation (see Figure 4 and 5 below).

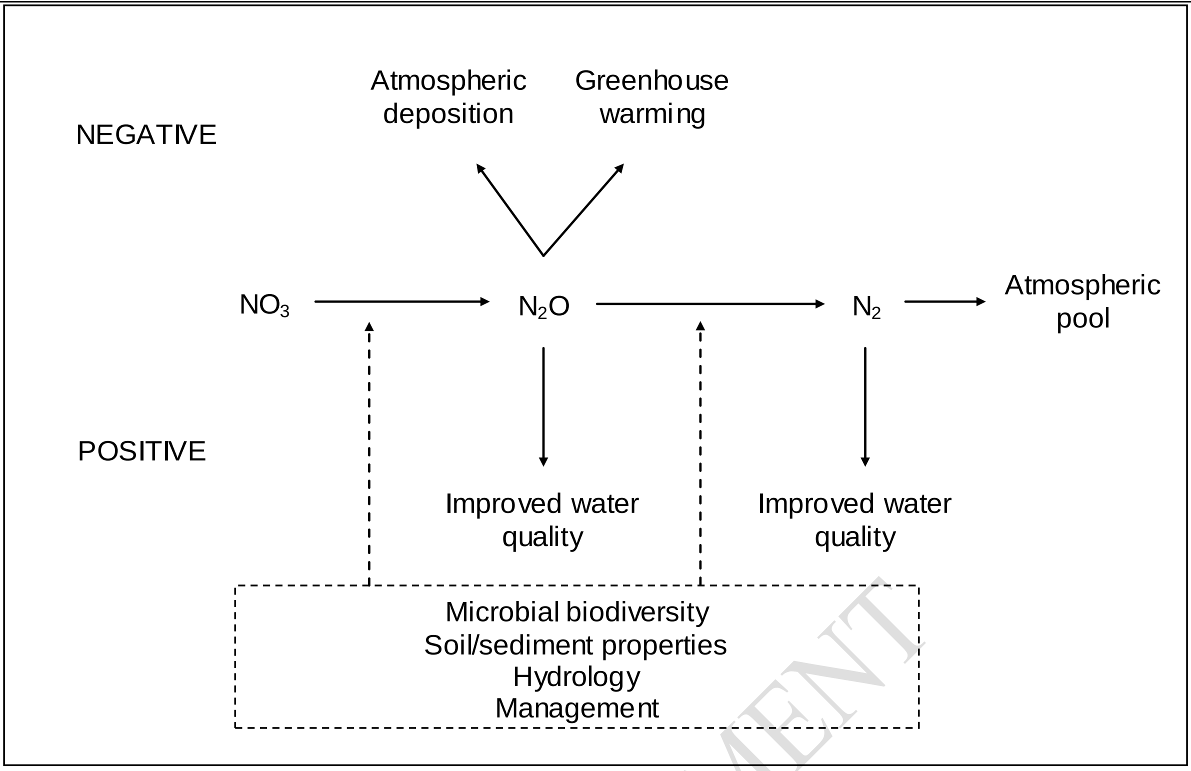

Figure 5: Wetland trade-off dilemma – water quality versus climate control

Many wetland soils support denitrifier microorganisms which convert nitrate to di-nitrogen gas in a two stage process with nitrous oxide as the intermediary product. This may result in a major service of water purification, protecting the biodiversity of adjacent waters (Barker et al. 2008). However the balance of N_2O or N_2 production, which is controlled by soil ecosystem properties, may generate a major disservice though release of a potent greenhouse gas (N_2O) as well as a potential source of atmospheric N deposition if full denitrification is not achieved.

3.3 Scales of provision

A number of key requirements need to be satisfied if knowledge of ecosystem services is to be effectively translated into operational practice. The need for a verifiable evidence base and understanding of the trade-offs resulting from interactions requires knowledge of the scale, and the temporal and spatial dynamics and distribution, of ecosystem service delivery. This will enable key questions to be addressed, such as ‘Where and to what extent are services being provided?’ and ‘How much of a particular ecosystem or individual component is necessary to deliver a particular service or combination of services?’

While some services may be realized on the same temporal or spatial scale as the system that generates them, notwithstanding the importance for ecosystem resilience of ecological connections with other ecosystems, others may be realized on completely different scales. These include pollination, which operates at a local scale and can be managed by ensuring that there are areas of land that maintain populations of pollinators in a mosaic of land-use types; hydrological services, which function at a catchment or river basin scale and which require co-operation among land managers at that scale; and carbon sequestration in soil organic matter, which operates at regional and global scales and necessitates policy decisions by governments and international bodies to ensure that appropriate incentives are in place to ensure necessary behaviour by local land managers.

Attempts at quantifying spatial aspects of multiple services include that of the service-providing unit (SPU), defined by Luck et al. (2003) as 'ecosystem structures and processes that provide specific services at a particular spatial scale'. For example, an SPU might comprise all those organisms contributing to pollination of a single orchard, or all those organisms contributing to water purification in a given catchment area (Luck et al. 2003; 2009). This is a parallel approach to that of the prediction of functioning and service provision in wetland ecosystems using a hydrogeomorphic unit (HGMU) approach (Maltby et al. 1994; Maltby 2009). The HGMU concept uses spatially-defined units to assess functioning at a range of landscape scales, and this method could feasibly be extended for assessment of other ecosystems. One of the major challenges in applying the SPU concept is to translate the unit into tangible and ideally mappable units of ecosystems and landscape/seascape, but the concept potentially offers an approach that focuses on multiple services and where changes to key species or population characteristics have direct implications for service provision.

4 Management of ecosystem services: dealing with uncertainty and change

In an increasingly globalized world, social conditions, health, culture, democracy, and matters of security, survival and the environment are interwoven and subject to accelerating change. Although change is inevitable, it is essential to understand the nature of change, especially the existence of thresholds and the potential for undesirable and, in practice, irreversible regime shifts. It is impossible to know where these potentially dangerous thresholds lie, and current efforts of adapting to climate change and other stressors will require a precautionary approach and a much deeper understanding of resilience and the combined capacity in both social and ecological systems if society is to cope with and benefit from change, i.e. social-ecological resilience (Folke et al. 2004). As the UN Secretary General has observed: "Building 'resilience thinking' into policy and practice will be a major task for all of the world's citizens throughout the new century" (UN Climate summit September 24, 2007).

4.1 Ecosystems, services and resilience

Physical influences on ecosystems include geology, climate, topography, hydrology, connectivity with other ecosystems, and the results of human activities. Frequent minor disturbances are characteristic of ecosystem functioning. Typically, these may be seasonal influxes of nutrients or organisms, variations in temperature or hydrology, or weather or age damage to structuring organisms e.g. trees, to which the resident species are adapted (e.g. Titlyanova 2009). Large and less frequent disturbances may follow geological disturbances, anthropogenic eutrophication (increased nutrient loading) or toxic pollution, habitat loss, disconnection from adjacent ecosystems, species invasions, climate change and other external drivers of ecosystem change (see later). These larger disturbances can drive permanent or long-term ecosystem change by altering the physical structure of the ecosystem, and through removal of species and alteration of species interactions. Grazing pressure from elephants, for example, can result in the long-term replacement of woodland by grassland (de Knegt et al. 2008). Human activities such as atmospheric deposition of chemicals (Vitousek et al. 1997; Phoenix et al. 2006) have ensured that no pristine environments remain on Earth (Lawton 1997).

Ecosystem responses to changes in key variables such as temperature, nutrient loading, hydrology or grazing may, for example, lead to an increase in productivity (e.g. Aberle et al. 2007), or alter competition between species (Rahlao et al. 2008) by tipping the balance in favour of one species over another as a function of the response diversity. This type of change may be reversible and in proportion to the degree of change in either direction. For example, eutrophication of a normally nutrient-poor ecosystem typically leads to increased production and greater species diversity until a point is reached of high nutrient loading, when fewer, robust species come to dominate (Grime 1988; Badia et al. 2008). Over-grazing or cultivation of a forested ecosystem might alter the vegetation by halting recruitment of new trees and by favouring herbaceous species tolerant of grazing and trampling (Mysterud 2006). If the eutrophication or grazing driver is removed, the site may revert to its former state, unless key species have been eliminated.

A given ecosystem state may be maintained during moderate environmental changes by means of buffering mechanisms. These are negative feedbacks, which maintain the prevailing ecosystem state by containing the potentially exponential growth of some species. For example, in shallow lowland lakes, the increase of fast-growing phytoplanktonic algae is controlled by competition from macrophytes (plants and large algae), which store nutrients, making them unavailable to the algae, and also by invertebrate zooplankton, which grazes the algae. These zooplpankters hide from predators in the macrophytes, while predatory fish such as pike (*Esox lucius*) lurk in the macrophyte beds to ambush the smaller fish that consume zooplankton. Together, these buffering mechanisms serve to reinforce the clear-water status of the lake (Scheffer et al. 1993).

The capacity of an ecosystem to withstand perturbations without losing any of its functional properties is often referred to as *ecosystem resilience*. In practice, minor disturbances to ecosystem stability can serve to increase resilience overall because they impose the necessity for flexibility on species interactions (Gunderson 2000), hence Holling’s original (1973) definition of the term as “the capacity of a system to absorb and utilize or even benefit from perturbations and changes that attain it, and so to persist without a qualitative change in the system". Westman (1978) described resilience as the ability of an ecosystem to recover from disturbance without human intervention. Today the most common interpretation of resilience is that it represents the capacity of a system (e.g. a community, society or ecosystem) to cope with disturbances (e.g. financial crises, floods or fire) without shifting into a qualitatively different state (Gunderson and Holling 2002). A resilient system has the capacity to withstand shocks and surprises and, if damaged, to rebuild itself. Hence, resilience is both the capacity of a system to deal with change and continue to develop (see also Brand 2005; Brand and Jax 2007).

Where environmental drivers are persistent or strong, ecosystems may pass a threshold and undergo sudden and catastrophic structural change (Thom 1969; Loehle 1989; Walker and Meyers 2004). This can shift the ecosystem to an alternative state (Holling 1973; May 1977; Scheffer et al. 2001), which is also sometimes termed a '*regime shift*' (Folke et al. 2004). Such regime shifts can produce large, unexpected, changes in ecosystem services. Examples at local and regional levels include eutrophication of lakes, degradation of rangelands, shifts in fish stocks, breakdown of coral reefs, and extinctions due to persistent drought (Folke et al. 2004). Environmental drivers may not instigate the regime shift directly, but may increase the susceptibility of the ecosystem to change following some disturbance. This has been elaborated particularly in shallow lake ecology (Irvine et al. 1989; Scheffer et al. 1993; Moss 2001). Continuing the lake example above, once an ecosystem is placed under stress from eutrophication, the loss of its macrophytes (perhaps through physical or chemical damage) paves the way for algal dominance, resulting in turbidity, because it removes the refuge of zooplankton from predation and the shelter needed by predatory fish that formerly kept zooplanktivorous fish numbers low. In addition, plant loss increases mixing and re-suspension, and removes competition for nutrients, leaving phytoplankton to dominate the ecosystem (Ibelings et al. 2007). Once these buffering feedbacks of the clear-water state are removed, the ecosystem is subject to the prevailing eutrophication driver, and new buffering feedback mechanisms reinforce the degraded ecosystem state.

Crucially, the clear and turbid states are alternatives under similar nutrient regimes. Each is held in place by its own buffering feedback mechanisms (Jeppesen et al. 2007). This means that a simple reduction in nutrient loading to the lake will not result in a reversion to its former clear water status. The change is thus said to be non-linear. Overcoming the buffering mechanisms is neither easy nor cheap nor often possible (see e.g. Phillips et al. 1999). Similar examples have been recorded in coastal systems (Palumbi et al. 2008) and in terrestrial field ecosystems (Schmitz et al. 2006) and woodlands (Walker et al. 1981) among others.

A hypothetical example is provided by a farm that uses most of its land for agriculture, providing a basic provisioning service. Assume that insectivorous birds provide a regulating service in preventing insect pest outbreaks. As long as bird abundances are high, a good crop is produced every year, given schematically as the upper line in Figure 6. If insufficient bird predation on insects occurs there will be a pest outbreak and crop production will be largely reduced (lower line). Since insect abundance is then too high for bird predation capacity (due to a non-convex feeding response function) there is an hysteresis effect, that is, there is a response delay, making it non-linear. The year to year abundance of all bird species that eat insects varies over time, yielding a certain probability distribution for total bird predation every year. The probabilities of crossing from one state into the other are given in Figure 6 as **a** and **b** respectively.

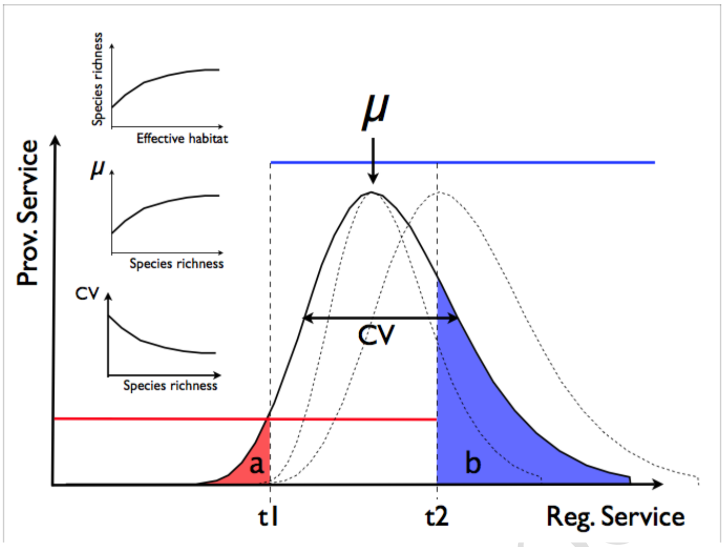

Figure 6: **Interactions between provisioning (crops) and regulating services (biological control).** Upper line –high generation of a biological control (insectivorous birds) leads to a high yield of crops. Lower line –decreased supply of biological control means a decline in crop yield. The characteristics of the larger landscape (inserted graphs) may give rise to non-linearities and thresholds (a and b) and the dynamics of the provisioning service being determined by management in the surrounding landscape (for details see text).

An important characteristic of this system is the shape of the probability function for the regulating service. Since this service is provided by a functional group of species, insectivorous birds, it is useful to examine how the species richness of this group of species would influence its performance. First, number of species may depend on the area of suitable habitat for birds, according to standard species-area relationships (upper inset graph). Species richness is generally positively correlated to total abundance due to complementary resource use among different bird species as shown in the middle inset graph. Higher species richness also results in lower coefficient of variation (CV) in the total bird abundance if there is some degree of negative autocorrelation between different species (Hughes et al. 2002). This negative autocorrelation can be thought of as species having different response functions and the effect on total bird abundance CV is shown in the lowest inset graph. The result of these effects is a change in the probability distribution of the regulating service, insect predation as shown as dashed lines. A consequence of increasing land available for natural bird populations is that the probability of falling into the undesired state with insect outbreaks (lower line) is largely reduced. At the same time the probability of recovering if this should happen increases. One consequence of such relationships is that the expected long-term value of the provisioning service largely depends on the management actions in the surrounding area.

4.1.1 Thresholds, recovery and ecological restoration

Although in many cases it may be possible to restore 'mildly' degraded ecosystems back to some defined earlier state, ecosystems can be so altered that restoration in the strict sense of the term is no longer possible. Given the extent of human impacts on the biosphere, the notions of restoration and restorability need to be expressed with many caveats.

The Society for Ecological Restoration International's *Primer on Ecological Restoration* (SER 2002) defines ecological restoration as 'the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed'. The target is a resilient ecosystem that is self-sustaining with respect to structure, species composition and function, while integrated into a larger landscape and congenial to 'low impact' human activities. However, if an ecosystem has been so disturbed that it crosses one or more thresholds, then restoration, if possible at all, may be achieved only with great difficulty and expense. Furthermore, subjective choices and trade-offs are inevitable and will be based on non-financial or marketable values as well as money and ecological science. For the restoration manager, the awareness of past threshold crossings in these cases should help guide the restoration programs (Clewell and Aronson 2007). Ease and cost will also vary among ecosystems, which impose the need for pragmatism. For example, in the Area de Conservación Guanacaste, in north-western Costa Rica, one of the longest running and largest scale restorations in the world, Janzen and co-workers have found that tropical dry forest recovers much more readily than nearby tropical moist forest (Janzen 2002). Budgeting of investment in labour and other expenses is allocated accordingly.

This example also highlights the fact that ecosystem resilience should not be interpreted in a normative way. Undesirable ecosystem states, such as cow pastures seeded with exotic grasses, may become very resilient to change, and restoration will need an understanding of how to erode undesirable resilient states. Managers should also not always seek to restore in terms of an original or 'pre-disturbance' ecosystem state or trajectory. Recovery of ecosystem *services* through revitalization of ecosystem *processes* may be better options. Placing emphasis on restoring *process* – and thus, ultimately, on ecosystem services – rather than on restoring a specific species inventory (Falk et al. 2006) also facilitates valuation and financing (De Groot et al. 2007; Holmes et al. 2007; see section 5.2 below). This is consistent with the first principle in applying the 'ecosystem approach' under the Convention on Biological Diversity (CBD 2000-2008).

4.2 Resilience thinking in policy and practice

In order to develop a deeper resilience thinking of relevance for policy and practice, there are at least three factors that need to be understood:

Depletion of non-renewable resources. A historic or geological legacy of environmental conditions may be supporting current patterns of use, but these are not being renewed on human time scales. Aside from minerals and fossil fuels, examples include groundwater resources and aquifers effectively containing 'fossil' water built up under different climatic conditions in previous millennia (Foster and Loucks 2006), and carbon storage in peat-based ecosystems developed since the end of the last glacial period, where net accumulation of peat no longer occurs. These represent resources that may be exhausted through use.

The changing environmental 'envelope'. The environmental envelope within which organisms evolve and patterns and processes of biodiversity and ecosystems develop, is changing because of human activity. This raises far-reaching questions of reliance on biodiversity and ecosystems and their viability when their base-line of environmental support and pressure of use is shifting. Coined by Pauly (1995) the related term "shifting baselines" highlights the fact that successive generations adjust to the state of the environment they find themselves in, i.e.an ecosystem state which is already degraded may be accepted as 'normal'. Jackson et al. (2001) collated worldwide data to demonstrate that in marine systems, humans have had such a prominent impact of successively reducing species richness for so long that previous levels of biodiversity are today difficult to imagine. Shifting baselines have major implications for the sustainability of resource use.

The effects of environmental shocks and disturbances. Both natural (e.g. climate, flood, fire, landslide) and human-induced (e.g. climate, sea-level change, deforestation, overgrazing, over-fishing, river-regulation, impoundment, pollution) impacts may be unpredictable and uncertain, producing major step changes, surprises and regime shifts. When faced with change through environmental stress, ecosystems may pass a critical threshold where the existing ecosystem structure collapses (Folke et al. 2004). Erosion of resilience may be an effect of variables such as human-induced environmental changes and loss of biodiversity.

A simple process-response schema is used to represent these relationships (Figure 7). It represents an ecosystem as a functional entity and a result of interactions between its structure and biological composition and processes.

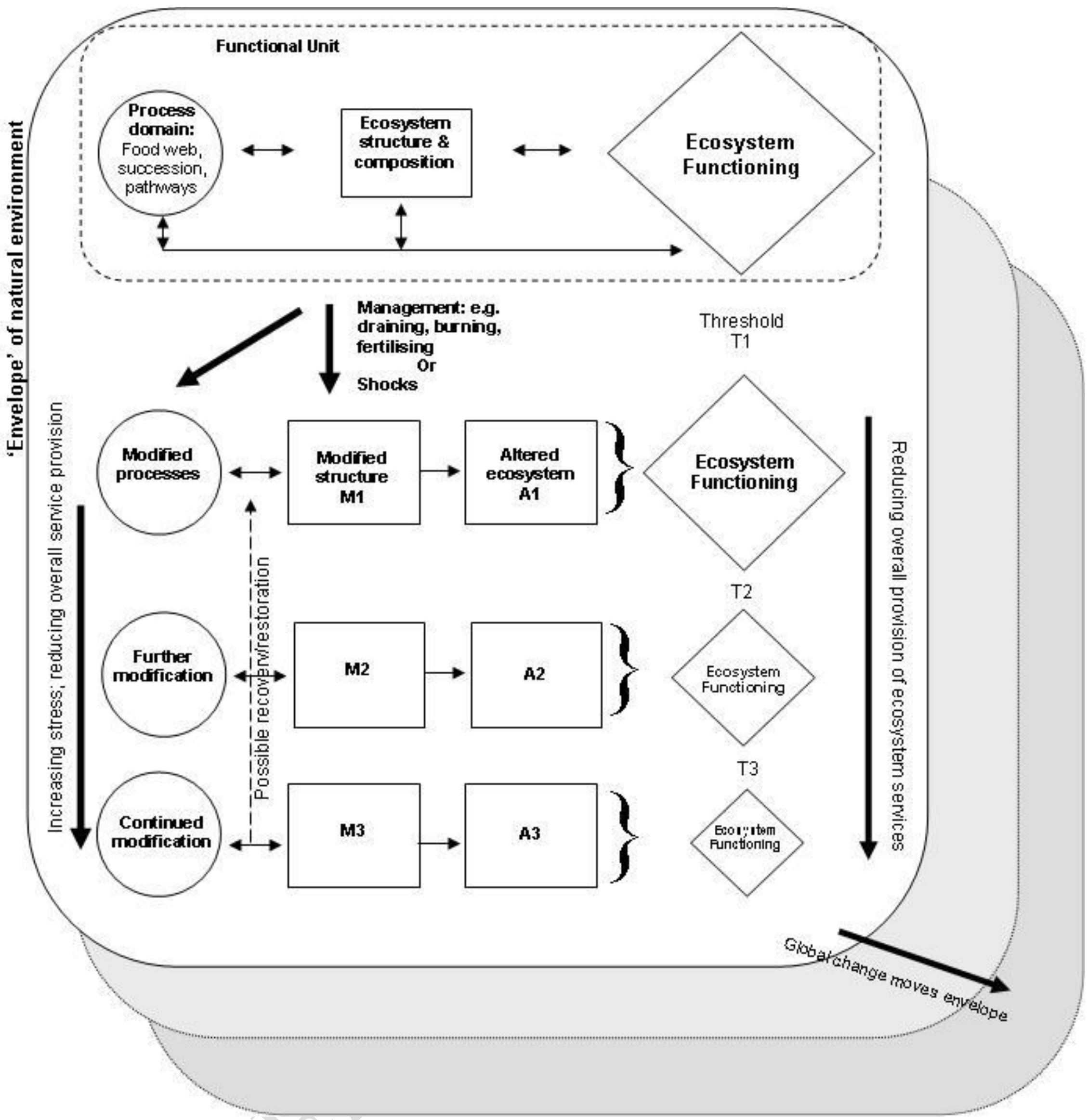

Figure 7: **A process-response model of ecosystem functioning and effects of impacts.** The changing environmental envelope of global environmental conditions alters the predictability of ecosystem responses to perturbations.

Source: Adapted from Maltby et al. (1994).

The ecosystem includes biological components representing particular genetic, species and community scale elements, all within a particular 'envelope' of prevailing environmental conditions. Both direct and indirect human influences as well as natural environmental shocks can modify ecosystem structure or the process domain, so altering ecosystem functioning. A sequence of increased modification resulting from increasing intensity of impact is shown as reducing the total combination of functions (although an individual function may actually increase, such as in the case of agricultural productivity). More or less distinct 'tipping points' may indicate the transition or shift from one state to another. The possibilities for recovery to previous ecosystem conditions will depend partly on external drivers discussed above, but mostly on the intrinsic properties and ecological integrity of the ecosystem, as well as time scale. However, erosion of resilience may make it impossible to recover a particular condition of functioning, structure or process because it no longer can occur within the boundary of the new envelope (i.e. due to changes in ecosystem dynamics).

This framework contests models and policies that are based on assumptions of linear dynamics, with a focus on optimal solutions. Applications of such theory and world views tend to develop governance systems that invest in controlling a few selected ecosystem processes, causing loss of key ecological support functions, in the urge to produce particular resources to fulfil economic or social goals (Holling and Meffe 1996). An alternative approach is based on ecosystems viewed as complex and adaptive, characterized by historical (path) dependency, non-linear regime shifts, and limited predictability. A dynamic view of nature and society has major implications for valuations of nature. If a system is discontinuous, the basic theorems of welfare economics are not valid and the result of resource allocation may be very far from the optimum, even if there are well-defined property rights (Mäler 2000). This has major implications for production, consumption and international trade, and also major implications for economic policy. Optimal management will often, because of the complex dynamics, be extremely difficult to implement.

5 Biodiversity, ecosystem services and human well-being

Since ecosystem services are the benefits that people get from ecosystems, it follows that changes in ecosystem services associated with changes in biodiversity will have implications for human well-being. Subsequent chapters explore the methods economists use to estimate the value of non-marketed ecosystem services, and summarize the results of existing valuation studies. Here we consider not how ecosystem services are valued, but how the production of ecosystem services confers value on all components of the biosphere, including biodiversity. The value of biodiversity derives from its role in the provision of ecosystem services, and from peoples' demand for those services. Economists have typically sought to value the individual components of ecosystems or specific services yielded by ecosystems, rather than ecosystems themselves. In some cases – where well-functioning markets exist – the valuation of specific services is straightforward. In most cases it is not. And even where markets for specific services do exist, derivation of the value of individual components of ecosystems is hard. The 'bundling' of services (4.1), and the fact that particular species may contribute to the production of many different services, mean that their marginal contribution to a particular service will be, at best, a partial measure of their value. In addition, the precise role of individual species in the ecological functioning that produces human benefits are typically known only for the well studied and highly controlled processes that yield marketed foods, fuels and fibres. Yet without this knowledge it is not possible to derive the value of either the basic building blocks of individual ecosystem services (however they might be defined) or of the functioning ecosystems that support an array of services. There may be numerous different pathways by which a particular function is generated and a resulting service delivered. Only exceptionally are all these pathways known – a challenge epitomized in the whole question of species 'redundancy'.

In principle, the value of ecosystems derives from the set of services – the discounted stream of benefits – they produce (Barbier et al. 2009; Barbier 2007). So if we define B_t to be the social benefits from the set of all services provided by an ecosystem at time t, then the present (discounted) social value of that system is:

$$V_0 = \int_0^\infty e^{-\delta t} B_t dt$$

where δ is the social rate of discount. For each time period, B_t is the sum of all benefits deriving from the ecosystem. That is, $B_t = \sum_i B_{it}$. Since those benefits depend upon sets of ecosystem services that, in turn, depend upon biodiversity, the value of of biodiversity can be derived from them. For example, if $B_t = f\left(S_t\left(X_t\right)\right)$ where S_t is the set of services produced, and X_t is the set of species, then the marginal value of the ith species is given by the derivative; $\frac{\partial B_t}{\partial S_t}\frac{\partial S_t}{\partial x_{it}}$. The main challenge for the calculation of the stream of benefits from ecosystems is that while a number of the services in an economy are marketed, many ecosystem services supported by biodiversity are not. Some benefits that contribute to human well-being do not have a price attached to them, and are therefore neglected in normal market transactions (Freeman 2003; Heal et al. 2005). Ecosystems are invariably multifunctional. Except in the case of managed systems (such as agro-ecosystems) their structure and function may be consistent with environmental conditions and historic patterns, but not the purposeful delivery of services. But even managed systems typically yield a range of benefits, and the value of the system depends not only on the value of the benefits that are the primary goal of management, but also on the ancillary services delivered as by-products of the primary services (Perrings et al. 2009).

How bundles of ecosystem services are configured matters. The beneficiaries of services may be spread among quite different stakeholder interests, and be distributed both off-site as well as on-site. Thus the value of freshwater quality improvement may be realized at various points downstream from the ecosystem performing the work and may be particularly significant even for the estuarine, coastal, and more distant marine waters and the services they, in turn, support.

Uncovering the value of ecosystem components requires an understanding of the ways in which they contribute to the production of ecosystem services. Neither individual species nor the ecosystems of which they are a part are exactly comparable or identical. The spectrum from microbial to charismatic top carnivore species encompasses a wide range of processes and functioning. The most charismatic species may be excellent indicators of ecosystem condition but not necessarily be of greatest functional importance. Ecosystem properties vary in detail according to both spatial and temporal factors. To derive the value of ecosystem components from their marginal impact on the production of valued goods and services, we need to know the shape of the ecological production functions that define the relationship between environmental inputs and outputs of goods and services. Ecological production functions thus capture the biophysical relationships between ecological systems and the services they provide, as well as inter-related processes and functions such as sequestration, predation, and nutrient cycling. They accordingly include both well-understood inputs over which humans have direct control, and poorly understood inputs over which humans have variable and often limited control. Identification of ecological production functions requires: i) specification of the ecosystem services of interest, and ii) development of a complete mapping from the structure and function of the ecological system to the provision of the relevant ecosystem services. Although we are making progress in understanding and defining ecological production functions for certain ecosystem services, such as carbon sequestration, the specification of production functions for many important ecosystem services is still rudimentary (Perrings et al. 2009).

Nevertheless, certain things are well understood. Conversion of ecosystems for the production of particular services generally reduces their capacity to provide other services. Whether specialization of this kind enhances the value of the ecosystem depends on the value of the forgone services (Balmford et al. 2002). Converted lands may gain value in terms of provisioning services, but lose value in terms of other types of services, such as water regulation, erosion control, habitat provision, fire regulation and so on. Conversion of natural forest to rice paddies and mangrove forest to shrimp ponds in many parts of South East Asia has led to a reduction in a range of regulatory functions, from storm buffering to silt entrapment (Barbier 2007). In the Mekong Delta, for example, acidification of potential acid sulphate soil materials, resulting from lowering of the water table and oxidation of the marine sediments, reduces the crop yield and harvest after just a few years, and often results in abandonment (Maltby et al. 1996). Such ecosystem alteration yields no obvious compensating gains. In such cases, the cost of reduced water quality, storm and flood protection, wildlife habitat and shrimp or fish recruitment from wild populations have still not been factored in to the decisions that lead to ecosystem change (Barbier 2007). Understanding and valuing the changes in the regulating ecosystem services involved is probably the biggest challenge to the economics of biodiversity at the moment.

The valuation of the contribution of biodiversity to regulating services poses particular challenges. The regulating services provide value through their role in assuring the reliability of service supply over space or time; sometimes expressed in terms of the resilience of the system to environmental shocks. That is, they moderate the variability or uncertainty of the supply of provisioning and cultural services. Estimation of the contribution of individual species to this service is, however, problematic. A small subset of species and their accompanying symbionts, mutualists, or commensalists supply the ecosystem services required for human survival. While an increase in biodiversity may increase production, experimental data indicate that for given environmental conditions this effect is small. The productivity of some biologically diverse communities, for example, has been found to be about 10% higher than the productivity of monocultures, but the effect often saturates at fewer than ten species. If environmental conditions are not constant, however, the effect may increase with the number of species providing that they have different niches and hence different responses to disturbances or changes in environmental conditions. For instance, the regulation of pest and disease outbreaks is affected by foodweb complexity. Simple predator-prey systems are prone to 'boom-and-bust' epidemiology, whereas the presence of multiple predators and prey operating at several trophic levels, and including prey-switching, involves much more stable dynamics (Thebault and Loreau 2006).

The value of biodiversity in regulating the provisioning and cultural services is illustrated in Figure 8. Since people care about the reliability (or variability) in the supply of these services (people are generally risk averse), anything that increases reliability (reduces variability) will be valued. The value of regulating services accordingly lies in their impact on the variability in the supply of the provisioning and cultural services. An important factor in this is the diversity of the functional groups responsible for the services involved. The greater the specialization or niche differentiation of the species within a functional group, the wider the range of environmental conditions that group is able to tolerate. In some cases greater diversity with a functional group both increases the mean level and reduces the variability of the services that group supports. Indeed, this portfolio effect turns out to be one of the strongest reasons for maintaining the diversity of functional groups.

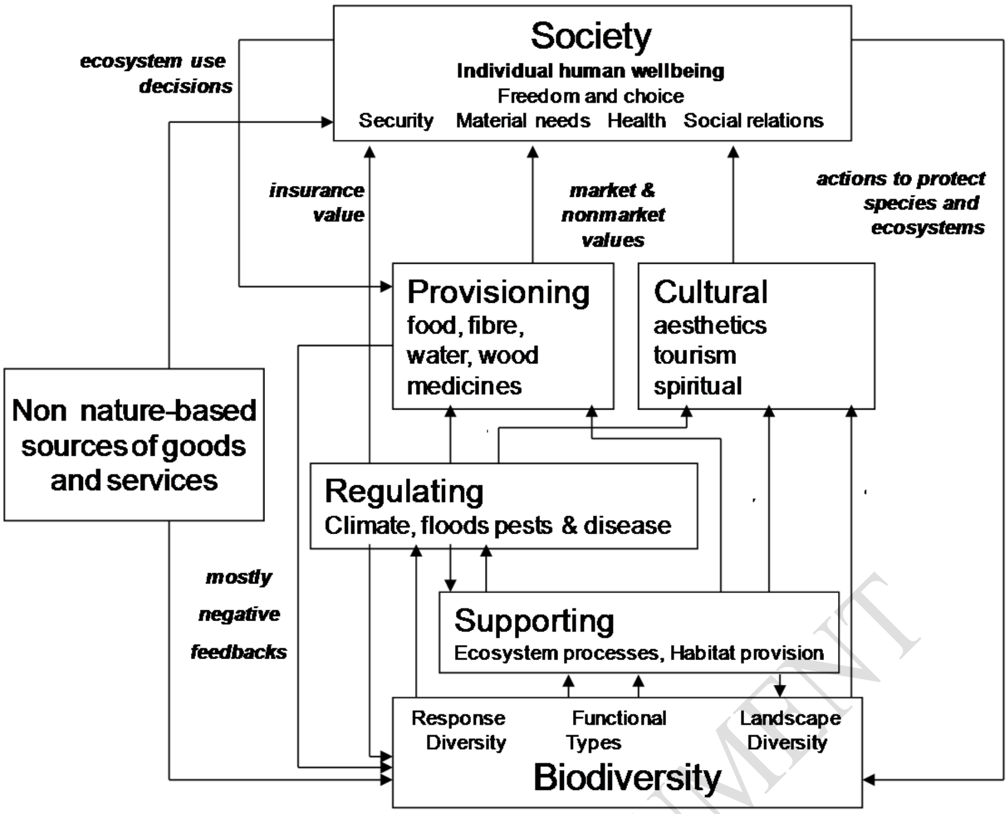

Figure 8: **Deriving the value of biodiversity and the regulating services.** Source: Kinzig et al. (2009).

The general point here is that the value of ecosystem components, including the diversity of the biota, derives from the value of the goods and services they produce. For each of the ecosystem services described in this chapter we have identified its sensitivity to changes in biodiversity. If greater diversity enhances mean yields of valued services it is transparent that diversity will have value. However, it is also true that if greater diversity reduces the variance in the yield of valued services that will also be a source of value. Since people prefer reliability over unreliability, certainty over uncertainty, and stability over variability, they typically choose wider rather than narrower portfolios of assets. Biodiversity can be thought of as a portfolio of biotic resources, the value of which depends on its impact on both mean yields and the variance in yields.

It follows that there is a close connection between the value of biodiversity in securing the regulating services, and its value in securing the resilience of ecosystems. Since resilience is a measure of the capacity of ecosystems to function over a range of environmental conditions, a system that is more resilient is also likely to deliver more effective regulating services. The economics of resilience are considered in more detail in Chapter 5.

The various forms of natural capital (see MA 2005) provide and regulate flows of ecosystem services essential to life and economic production (de Groot et al. 2002). However, with the exception of agricultural land, it is currently undervalued, and sometimes even invisible, both in our national and international systems of economic analysis, and in indicators like Gross Domestic Product (GDP) (Arrow et al. 1995; Dasgupta et al. 2000; TEEB 2008; TEEB 2009). Furthermore, global society is making withdrawals of natural capital stocks far in excess of its yield in interest (ecosystem services, e.g. carbon sequestration) and societal re-investments in natural capital are, to date, limited. Yet increasing evidence shows that investing in the restoration and replenishment of renewable and cultivated natural capital 'pays' in economic and political terms (e.g. Goldstein et al. 2008).

As shown in chapter 1, the concept of restoring natural capital is a relevant concept defined as all investments in renewable and cultivated natural capital stocks and their maintenance in ways that improve the functions of both natural and human-managed ecosystems, while contributing to the socio-economic well-being of people (Aronson et al. 2007). It is thus a broader concept than ecological restoration (section 5.1.1), can be applied at landscape or regional scales, and can generate significant economic savings. For example, integrated programs seeking to restore degraded natural systems and rehabilitate production systems in the Drakensberg Mountain Range Project (Blignaut et al. 2008).

6 Conclusions and further research

We now have a good understanding of the intricacies and outcomes of ecological dynamics as well as the expressions of these processes in the provision of goods and services to human society. Significant gaps in our knowledge remain, but there is an emerging scientific consensus on the need to sustain biological diversity to protect the delivery of ecosystem services. Nevertheless, if we wish, for example, to predict the impact of biodiversity change on variability in the supply of ecosystem services, we need to measure the impact of biodiversity conservation over a range of environmental conditions. In the same way, we need to be able to identify the effect of biodiversity change on the capacity of social-ecological systems to absorb anthropogenic and environmental stresses and shocks without loss of value (Kinzig et al. 2006; Scheffer et al. 2000; see Figures 8 and 9). To do so, requires analysis of the linkages among biodiversity change, ecological functioning, ecosystem processes, and the provision of valued goods and services. To understand and even enhance the resilience of such complex, coupled systems, therefore, we need robust models of the linkages between biodiversity and ecosystem services, and between biodiversity change and human well-being (Perrings 2007).

A major gap in knowledge is how different ecosystems interact in the delivery of services. Ecosystems are rarely homogeneous. Extensive forest ecosystems often contain rivers, lakes and wetlands as well as patches of land which may be farmed or managed as open habitat for wildlife. It is important to know how the various combinations of ecosystems operate together to generate services, which may be enhanced or impeded by interactions. Equally we need urgently to develop evidence-based management practices that maximize the delivery of a broad range of services from individual ecosystems, especially where these are managed intensively for food or fuel production.

Existing knowledge is also sufficient to develop more effective instruments for ecosystem service-based biodiversity conservation, including the payments for ecosystem service (PES) systems discussed elsewhere in this report (TEEB 2009). These instruments offer a mechanism to translate external, non-market values of ecosystem services into real financial incentives for local actors to provide such services (Ferraro and Kiss 2002; Engel et al. 2008). Similarly, existing knowledge is sufficient to develop more effective governance institutions, including property rights regimes and regulatory structures. Once such mechanisms are established, their effectiveness can be increased by improving the quality of available information on the effect of conservation on ecosystem service provision.

Major questions that future research needs to address include:

1. Understanding links between biodiversity, ecosystems and resilience:

- What are the roles of species interactions and functional diversity for ecosystem resilience?
- What are the drivers behind loss of resilience and how do they interact across scales?
- What are the impacts of climate and related environmental changes on ecosystem functioning through effects on species (re)distribution, numbers and process rates?

2. Understanding the dynamics of ecosystem services:

- How can we better analyze effects on regulating ecosystem services of an increase in provisioning services?
- What tools can contribute to accurate mapping of land and seascape units in terms of functioning and service support / provision?
- What specific tools could contribute to better assessment of spatial and temporal dynamics of service provision, especially in relation to defining who benefits, where and to what extent?

3. Understanding the dynamics of governance and management of ecosystems and ecosystem services:

- If all ecosystem services are taken into account, what is the appropriate balance between ‘more diverse landscapes generating bundles of ecosystems services’ and more intensively managed ecosystems like monocultures for food production?
- What are the trade-offs and complementarities involved in the provision of bundles of ecosystem services, and how do changes in the configuration of ecosystems affect their value?
- What are the most effective mechanisms for the governance of non-marketed ecosystem services, and how can these be designed so as to exploit future improvements in our understanding of the relationships between biodiversity, ecosystem functioning and ecosystem services?

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Chapter 3

Measuring biophysical quantities and the use of indicators

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Contents

Key messages

- A lack of relevant information at different scales has hampered the ability to assess the economic consequences of the loss of ecosystems and biodiversity.
- Most of the current measures and indicators of biodiversity and ecosystems were developed for purposes other than the economic assessment outlined by TEEB. They are therefore unable to show clear relationships between components of biodiversity and the services or benefits they provide to people, making them less relevant to the audience and aims of TEEB.
- A reliance on these existing measures will in all likelihood capture the value of only a few species and ecosystems relevant to food and fibre production, and will miss out the role of biodiversity and ecosystems in supporting the full range of benefits, as well as their resilience into the future.
- A set of indicators is needed that is not only relevant and able to convey the message of the consequences of biodiversity loss, but must also be based on accepted methods that reflect the aspects of biodiversity involved and the service that is of interest, capture the often non-linear and multi-scale relationships between ecosystems and the benefits that they provide, and be convertible into economic terms.
- While it is possible to obtain preliminary estimates of the consequences of biodiversity and ecosystem loss using existing data and measures, these must be complemented with active research and development into the measurement of biodiversity and ecosystem change, their links to benefit flows and the value of these flows so as to realize the full value of biodiversity and ecosystem management

1 Introduction

1.1 Aim and scope of this chapter

Changes in biodiversity, ecosystems and their services ultimately affect all people (MA 2005b). Global declines in biodiversity and ecosystems, the ongoing degradation and unsustainable use of ecosystem services, and the resultant effects on human wellbeing have led to many international and national responses focussed on halting and reversing this trend (Balmford et al. 2005).

However, attempts to halt or reverse these declines in ecosystems and biodiversity are confounded by a lack of information on the status and changes in ecosystems and biodiversity, the drivers of change, and the consequences of management responses (Pereira and Cooper 2006). The information that does exist remains fragmented, not comparable from one place to another, highly technical and unsuitable for policy makers, or simply unavailable (Scholes et al. 2008; Schmeller 2008).

Over the past decade, several programs have sought to fill some of these information gaps, from local to global levels (Royal Society 2003; Pereira and Cooper 2006; Scholes et al. 2008). The purpose of TEEB and this chapter is twofold: to provide guidance to interested stakeholders on the strengths and weaknesses of available measures and indicators of biodiversity and ecosystem status and change, with a focus on those which can put an economic value on these changes (TEEB 2008); and to outline what is needed to improve the existing science base of biodiversity and ecosystem indicators to better meet the needs of TEEB and associated efforts.

The chapter also describes in detail a set of global and sub-global indicators to highlight the opportunities and challenges associated with developing indicators which can be used in assessing the economic consequences of changes in biodiversity and ecosystems.

1.2 Why are indicators needed?

Ecosystem and biodiversity indicators serve multiple purposes which can broadly be categorized into three key functions: (1) tracking performance; (2) monitoring the consequences of alternative policies; and (3) scientific exploration (Failing & Gregory 2003). This chapter will focus mostly on the first two roles. Indicators are defined here as variables indicating something of interest or relevance to policy- or decision-makers with some logical connection to the object or the process being measured. They reflect, in an unambiguous and usually quantitative way, the status, causes (drivers) or outcome of the process or object (Ash et al. 2009). Indicators simplify and quantify information so that it can be easily communicated and intuitively understood, allowing policy- and decision-makers to base their decisions on evidence (Layke 2009).

It is useful to distinguish between measures, indicators and indices, the key terms used in this chapter. The term *measure* (or measurement) is used to refer to the actual measurement of a state, quantity or process derived from observations or monitoring. For example, bird counts are a measure derived from an observation. An *indicator* serves to indicate or give a suggestion of something of interest and is derived from measures. For example bird counts compared over time, show a trend which can indicate the success of conservation actions for a specific group of species. Indicators are typically used for a specific purpose, e.g. to provide a policy maker with information about progress towards a target. An *index* or multiple *indices* are comprised of a number of measures combined in a particular way to increase their sensitivity, reliability or ease of communication. These are useful in the context of biodiversity assessment where multiple attributes and measurements, related to a wide variety of policies, have resulted in long lists of measures and indicators. To communicate these trends in a small number of simple and meaningful indices is sensible (Balmford et al. 2005). For example, in the Red List Index for birds, changes in threat status over time are expressed as a number, obtained through a specific formula. A concern with composite indices is that the underlying measures often become obscured. Ideally they should be disaggregateable and traceable back to the original measures (Scholes and Biggs 2005).

1.3 What makes a good indicator?

The Convention on Biological Diversity (CBD 2003), as well as a number of other publications (Royal Society 2003; Mace and Baillie 2007; Ash et al. 2009), list multiple criteria to consider when selecting and developing indicators and measures of ecosystems and biodiversity. Of these criteria, perhaps the most pertinent to this chapter and its readers is the need to make the indicators relevant to the purpose. This not only requires setting clear goals and targets in the indicator development process, but also a thorough understanding of the target audience and their needs (Mace and Baillie 2007).

Vagueness in current targets, the diversity of target audiences and their needs, the resources required to turn measures into effective indicators, and the reliance of most current measures and indicators on available data have posed substantial obstacles in the development of relevant and useful indicators (Royal Society 2003; Green et al. 2005; Mace and Baillie 2007; Layke 2009).

Much of the current effort in indicator development has arisen from the CBD's Biodiversity 2010 Target and regional or national responses to this target (e.g. EEA 2009), as well as the work of the Millennium Ecosystem Assessment (MA: MA 2005b). While the latter did not aim to develop indicators, the global and sub-global assessment of ecosystem status and trend collated many measures of ecosystems and ecosystem services (Layke 2009). Both of these initiatives have resulted in substantial effort and resources invested in indicator development and the collation of measures, with good progress in some aspects of the assessment of biodiversity, ecosystem and ecosystem service status and trends (Mace et al. 2005; Mace and Baillie 2007; EEA 2009; Layke 2009; http://twentyten.net; http://www.unep-wcmc.org/collaborations/BINU/). However, many gaps and substantial challenges remain for scientists and policy makers in ensuring that the measures and indicators are sensitive, realistic and useful (MA 2005b; Mace and Baille 2007; Scholes et al. 2008; Layke 2009).

In the context of TEEB it is important to recognize that its objectives and audience differ from existing programs like the Biodiversity 2010 Target and the MA. TEEB moves beyond the measurement of biodiversity and ecosystem status and change, to an assessment of the economic implications of changes in biodiversity and ecosystems (TEEB, 2008). It is therefore possible that existing indicators and measures developed mainly for the Biodiversity 2010 Target and for the MA's purposes, may not best address the objectives of TEEB.

The intended audience of TEEB is wider and more varied than previous biodiversity and ecosystems indicator program audiences and comprises stakeholders at different levels, including individuals whose livelihoods directly depend on harvesting natural resources, resource managers, decision makers at all levels, and civil society in general. The scientific community is also a stakeholder as scientists are involved in the monitoring and observation of a broad range of biodiversity and ecosystem measures over a variety of scales (Schmeller, 2008). This varied audience will require different sets of indicators relevant and understandable within and across sectors and scales. Taking a sectoral perspective also implies combining measures that provide a broad integrated time series of ecosystem status at the relevant scale with relevant socio-economic indicators related to issues such as employment or trade.

TEEB's focus on the economic consequences of changes in biodiversity and ecosystems brings with it new challenges to the science and practice of biodiversity and ecosystem indicator development. First, TEEB is interested in the measurement of *biodiversity change*. This is a concept with which the Biodiversity 2010 Target indicator development has struggled. Not only is biodiversity a multi-faceted, multi-attribute concept of a hierarchy of genes, species and ecosystems, with structural, functional and compositional aspects within each hierarchical level (Noss 1990). Change in biodiversity is also multi-faceted and can include loss of quantity (abundance, distribution), quality (ecosystem degradation) or variability (diversity of species or genes) within all levels and aspects (Balmford et al. 2008). As Mace et al. (2005) highlight, different facets of change will have different implications for different ecosystem services, for example changes in functional and structural variability in species will have broad-ranging impacts on most services, while changes in the quantity and distribution of populations and ecosystems will be important for many provisioning and regulating services. Therefore the most appropriate measures and indicators will involve a consideration of both the aspects of biodiversity involved and the service that is of interest.

Second, in order to assess the consequences of change in biodiversity TEEB is targeted at the *links* between biodiversity, ecosystem services and human wellbeing. While there may be good progress in the development of indicators to measure the status and trends of biodiversity, ecosystem services and human wellbeing, TEEB needs measures that can capture the often non-linear and multi-scale relationships between ecosystems and the benefits that they provide (van Jaarsveld et al. 2005; Fisher and Turner 2008). This is an area of very little current development and investment, especially at global scales.

Finally, TEEB is interested in the *economic* consequences of biodiversity change. Therefore indicators and measures used in TEEB must be convertible into economic terms and suitable for economic analyses. This implies more than the generation of monetary values, and requires the inclusion of livelihood conditions, risk, and access to resources, benefit sharing and poverty considerations (Balmford et al. 2008) (see Chapters 4 & 5). Since TEEB's ultimate aim is to make the use of natural resources more sustainable, indicators should address the sustainability of the use patterns measured. TEEB, although acknowledging the importance of nature's intrinsic worth, does not explicitly address intrinsic values of nature, including the ethical considerations regarding the rights of all species. At this stage, TEEB also does not cover the economic value of the interactions between species that structure ecological processes, though this is relevant to the assessment of ecosystem services and may be attempted in future.

This chapter aims to take these challenges into account and through an assessment of existing measures and indicators to identify which of the available measures are the most appropriate for the purposes of assessing the economic consequences of biodiversity and ecosystem change. In this context, good measures would be measured with known precision and should sample across relevant places or systems, they ideally would be repeatable and have a history, and would have a clear relationship to some benefit that people receive from biodiversity or ecosystems.

In addition to these general characteristics, indicators and measures need to have an appropriate temporal and geographical coverage, and ideally be spatially explicit.

The importance of being spatially explicit has been emphasised in TEEB by Balmford et al. (2008). The production, flow and use of the benefits of biodiversity and ecosystems varies spatially, as do the impacts of policy interventions. Making available data and information spatially explicit helps make assumptions explicit, and also identifies needs for further information. The production and use of the benefits of ecosystems and biodiversity often take place in different geographical areas, so a spatially explicit approach is essential to fully evaluate the importance of ecosystem services and the impacts of related policy actions.

2 Existing measures and indicators

Biodiversity, ecosystem and ecosystem service indicators and measures have proliferated over the past several years, largely in response to the setting of the CBD Biodiversity 2010 Target and the Millennium Ecosystem Assessment and its sub-global activities. An exhaustive review of all these indicators and measures is not intended here (see Mace and Baillie 2007; Layke 2009 for in depth reviews of indicator groups); rather this section highlights what types of indicators and measures are available and reviews their relative strengths and weaknesses in an effort to guide the selection and development of appropriate indicators and measures that can be used to assess and predict the economic consequences of biodiversity and ecosystem change.

Biodiversity and the ecosystems that it structures are notoriously complex entities to measure and assess, and this can be undertaken in a variety of different ways. The MA (2005a) and Balmford et al. (2008) highlighted that biodiversity indicators are available for assessing all the different levels of the biodiversity hierarchy (genes, species, ecosystems), as well as measuring several attributes at these levels, namely *diversity, quantity* and *condition*. These three categories of attributes are used to structure this review and Table 1. A fourth category of indicators is one that measures *pressures* exerted on the environment. This chapter also includes an additional category focussed on *ecosystem service* measures and indicators, in recognition of the large amount of data and measures made available through the MA and its follow up activities, as well as the importance of these measures in linking biodiversity to economic valuation. The ecosystem service measures are separated into *provisioning, regulating and cultural service* categories due to the different relationships between these groups of services and ecosystem elements (see chapter 2), as well as the different tools available for valuing different ecosystem service groups (see chapter 5). TEEB (2009, chapter 3) provides a list of examples of ecosystem service indicators, but this review focuses on only those measures and indicators which are already in use and thus available for review.

This chapter does not propose a specific set of indicators and measures; as discussed earlier, different sets of indicators will be required for different audiences. Rather, the chapter provides an overview of existing indicators and measures of biodiversity and ecosystems and their potential use in economic valuation exercises like those adopted by TEEB. The chapter focuses on existing spatially explicit indicators and measures (with some mention of those known to be in development). This chapter assesses their current application in biodiversity and ecosystem service measurement and in valuing change, their ability to convey information and their data availability. These last two criteria were developed and applied in the WRI review of the MA measures and indicators (Layke 2009). In this review indicators are ranked based on their ability to convey information as a combination of their intuitiveness, sensitivity and acceptability, and their data availability based on the presence of adequate monitoring systems, availability of processed data and whether the data are normalized and disaggregated. This chapter does not provide an evaluation of use, access or human wellbeing indicators.

Table 1: Review of existing biophysical measures in terms of their application to measuring biodiversity and ecosystems, their ability to convey information and current data availability at the global scale

Broad category of origin	Category	Examples	Application	Ability to convey information	Data quality and availability
Biodiversity measures and indicators	Measures of diversity	Species diversity, richness and endemism Beta-diversity (turnover of species) Phylogenetic diversity Genetic diversity Functional diversity	To biodiversity: These measures are used to identify areas of high biodiversity value and conservation priority at global and sub-global scales. Seldom used to measure change at global scales, but have been used to indicate functional and structural shifts associated with declines in diversity at sub-global scales. Trends in genetic diversity of species is a Headline Indicator (HI) for Biodiversity 2010 Target To ecosystem services: Not easily linked to specific provisioning or regulating ecosystem services, with the exception of proposed measures of functional diversity. Analysis of congruence between diversity and service levels shows mixed support. Studies demonstrate importance of species and	Measures and maps of areas of high species diversity and endemism easily understood by wide audience, based on agreed methods and data. Not sensitive to short term change	Species measures for some taxa available globally, but not as a time series Other measures not available globally

			genetic diversity in promoting ecosystem resilience across ecosystem services. Genetic diversity also linked to options for bio-prospecting and food security. Cultural values of diversity, especially education, research and aesthetic values, provide these measures with a link to cultural ecosystem services. To valuation: Not easily valued due to general rather than specific role in providing benefits. Some valuation of bio-prospecting and genetic diversity of crop species possible. Also possible to value the cultural values attached to diversity, although not yet common practice.		
	Measures of quantity	Extent and geographic distribution of species and ecosystems Abundance / population size Biomass / Net Primary	To biodiversity: Descriptive measure of biodiversity used in baseline studies and descriptions; when available over temporal scales they can feed into indicators of biodiversity status and trends, and prioritisation and risk assessment protocols. Trends in selected ecosystems and species are HIs of the Biodiversity 2010 Target To ecosystem services: Measure of status and	Measures and indicators of trends in habitat area and species populations are intuitive to a wide audience (e.g. deforestation rates). Measures of biomass and NPP	Global datasets of broad ecosystems and some taxa available for a single time period. For some species and populations there are good time series data.

		Production (NPP)	trends for ecosystems (e.g. forest, wetlands, corral reefs) and species (medicinal plants, food) which have clear links to provisioning services have been used as measures of stocks and flows of ecosystem services. Similarly useful for ecosystems and species with social and cultural values which have links to cultural services. Some use in measuring regulating services which rely on biomass or a particular habitat / vegetation cover (e.g. carbon sequestration, pollination, erosion control, water flow regulation). To valuation: Measures of provisioning, cultural and regulating services can be valued using the variety of approaches listed in Chapter 5 (e.g. market price, contingent valuation, factor income or replacement cost).	less intuitive. Most measures are based on accepted methods and are sensitive to change (data dependent)	NPP and Biomass measures available at global scales and can be modelled over multiple time series
	Measures of condition	Threatened species/ecosystems Red List Index (RLI) Ecosystem connectivity/fragmentation (Fractal dimension, Core	To biodiversity: These measures are used to assess and indicate the status and trends of biodiversity and ecosystems. Change in status of threatened species, Marine Trophic Index, connectivity/fragmentation, human induced ecosystem failure are HIs of Biodiversity	Threatened species status, RLI and MTI used and understood indicators of biodiversity loss,	Threatened species status and trends available for limited taxa at a global scale. Most other

		Area Index, Connectivity, Patch Cohesion), Ecosystem degradation Trophic integrity (Marine Trophic Integrity - MTI), Changes in disturbance regimes (human induced ecosystem failure, changes in fire frequency and intensity) Population integrity / abundance measures (Mean Species Abundance - MSA, Biodiversity Intactness Index -BII, Natural Capital Index- NCI)	2010 Target To ecosystem services: While providing an indication of the status and trend of ecosystems and their services, these indicators are seldom linked to quantified changes in ecosystem service levels. They are however useful indicators of sustainability, thresholds and the scale of human impacts on ecosystems, particularly where clear and demonstrable linkages exist. To valuation: Not currently converted into monetary values, although potentially useful in determining risk of economic loss	based on acceptable methods and data and sensitive to change. Other measures less intuitive and quite technical, little consensus on methods and data	measures only available at a sub-global scale and often only for one period of time.
	Measures of pressures	Land cover change	To biodiversity: These are measures of the pressures or threats facing biodiversity. They	Many of these measures and	Land cover data available at global

		Climate change Pollution and eutrophication (Nutrient level assessment) Human footprint indicators (e.g. Human Appropriated Net Primary Productivity - HANPP, Living Planet Index -LPI, ecological debt) Levels of use (harvesting, abstraction) Alien invasive species	do not measure the status and trends of biodiversity, but are an indication of the size and trends of the pressures on biodiversity and often feed into biodiversity assessments at national scales in State of Environment Reports. They are frequently used in communicating biodiversity status and trends and many are relevant to Biodiversity 2010 target To ecosystem services: When linked to particular species (e.g. fish) or ecosystems (e.g. wetlands) which provide or support ecosystem services, these measures are useful indicators of ecosystem service levels and declines. They are also used to indicate the sustainability of ecosystem service use and supply To valuation: Changes in ecosystem service levels lend themselves to valuation of the losses or gains in services. If information is available on threshold effects for particular services then these indicators can be useful in determining economic risk.	indicators are used to communicate the status of biodiversity to a wide audience (public and policy), consensus methods are in development, most are sensitive to change Composite footprint indicators are increasingly disaggregateable	scales, but not as a time series. Climate change models are globally available for a range of future time periods, linking these pressures to biodiversity changes remains a gap. Some measures of pollution available globally and over time (e.g. nitrogen deposition). Composite footprint indicators available globally and over time periods.

					Use levels and alien species under development
Ecosystem service measures & indicators[i]	Provisioning service measures	Timber, fuel and fibre production Livestock production Fisheries production Wild animal products Harvested medicinal plants Water yield and regulation Biological infrastructure needed for nature based recreation	To biodiversity: Measures of provisioning services currently used to indicate use and sustainability of use on biodiversity and ecosystems. More recently used to indicate the value of biodiversity and ecosystems To ecosystem services: Direct measures of ecosystem service levels and changes. When calculated as sustainable production measures can be used as indicators for monitoring and managing ecosystem services, contrasting sustainable production with actual. To valuation: Most indicators expressed as biophysical units which can be converted into monetary values where markets exist.	Simple and compelling indicators where they do exist. Methods of modelling and development not yet agreed upon. Sensitive to change	Timber and livestock production available globally Most data only available at sub-global scales and for single time period. Possibility of upscaling and modelling for some (see Section 3.3) Total production and direct use values more common than sustainable production

					indicators
	Regulation service measures	Carbon sequestration Water flow regulation and production Air quality regulation Natural hazard regulation Waste assimilation Erosion regulation / soil protection Disease regulation Pollination Maintenance of genetic diversity Pest control	To biodiversity: Many of these measures of measurements of ecological processes important to the persistence of ecosystems and so can be used to indicate functional biodiversity condition and trends. Recently used to indicate the value of biodiversity and ecosystems To ecosystem services: Direct measures of ecosystem service levels and changes. To valuation: Regulating services are more difficult to value but see Chapter 5 for progress in valuing through avoided / replacement or restoration and other costs. Double counting remains an issue with some of these services.	Less intuitive to a wide audience than the provisioning measures, excluding water and carbon which are increasingly understood. Limited consensus on methods of measurement and modelling. Less sensitive to short term changes	Most measures only at sub-global scales, although many identified as possible global indicators for development Carbon sequestration available globally Where data exist possible to model over time, but not common
	Cultural service	Recreational use Tourism numbers or	To biodiversity: Many of these measures are specific to particular ecosystems or species of	No such measures yet available	Most measures only at sub-global

	measures	income Spiritual values Aesthetic values	cultural value, although tourism can often be linked to habitat and species diversity. More recently been suggested as indicative of the value of biodiversity and ecosystems To ecosystem services: Direct measures of ecosystem service levels and changes. To valuation: Most cultural services are poorly understood and often difficult to value . Tourism and recreation services, as well as existence value more amenable to valuation. Some debate over the economic valuation of spiritual and religious values. See Chapter 5 for progress in valuing.	globally. At sub global levels some measures intuitive e.g. tourism numbers or recreational values. Other measures poorly understood. No consensus on measurement and modelling. Not sensitive to change	scales, although tourism identified as possible global indicator for development

An examination of Table 1 shows that there are a large number of measures and indicators available across geographic scales and regions for assessing biodiversity and ecosystem services. As in previous reviews of measures and indicators of biodiversity and ecosystem services (Royal Society 2003; Mace and Baillie 2007; Layke 2009), much of the existing data and indicators were collected and developed for purposes other than the one TEEB is interested in and are therefore not necessarily the right measures for assessing the economic consequences of biodiversity and ecosystem change. Furthermore most of the existing indicators are developed and applied within specific contexts resulting in some good biodiversity indicators and some progress in the development of ecosystem service indicators, but the current lack of measures and indicators which span contexts and show clear relationships between components of biodiversity and the services or benefits they provide to people is a key gap, making existing measures and indicators less relevant to the audience and aims of TEEB. The categories of indicators presented in Table 1 are reviewed below with two objectives: 1) to identify existing measures useful for economic valuation in the short term; and 2) to highlight the work still required to develop key fit-for-purpose indicators in the longer term.

2.1 Indicators of diversity

At a global level, measures and maps of species diversity, endemism and richness are available for some taxa e.g. mammals and amphibians (Myers et al. 2000; MA 2005a), while at sub-global scales these are supplemented by measures and indicators of genetic and ecosystem diversity (e.g. Bagley et al 2002). Although these indicators are the focus of many conservation agencies and policies and good at conveying their message of high biodiversity value, these measures are seldom used to assess the benefits provided by the diversity of genes, species, and ecosystems to people and economies. This is probably a result of the complex and tenuous relationships between diversity and ecosystem services (Balvanera et al. 2001; Hooper et al. 2005; Mace et al. 2005). While some evidence exists that diversity is important in resilience and adaptive capacity of biodiversity and ecosystems (Johnson et al 1996; Naeem 1998; Swift et al. 2004; Balvanera et al. 2006; Diaz et al. 2006), this “insurance value” is seldom calculated (see chapter 2 for a detailed explanation). A frequently cited example of the importance of diversity is the value of genetic diversity in agriculture and bio-prospecting (Esquinas-Alcázar 2005); however, these benefits are complex given their option-based nature and are therefore hard to quantify and value. Other benefits of diversity include cultural services associated with enjoyment and appreciation of diversity which may be more amenable to valuation, using for example willingness to pay approaches (e.g. Esquinas-Alcázar 2005), but the reliability of these approaches has not been demonstrated (see Chapter 5). Finally, functional diversity (i.e. the diversity of functional groups or types) is said to be important for regulating services (Bunker et al. 2005; Diaz et al. 2006, Chapter 2), but challenges with developing indicators of functional diversity, as well as the challenges associated with valuing regulating services (Chapter 5), limit the numbers and application of these indicators in valuation assessments.

These measures of diversity have good potential to convey a message in that they are intuitive, already widely in circulation, and largely based on accepted and rigorous methods and data. However, their sensitivity to change over policy relevant periods is weak because of data gaps, and because change would require local or global extinctions of species or ecosystems, which as Balmford et al. (2003) point out, is often a longer-term process insensitive to short-term change.

With the exception of genetic diversity, this category of measures is not the current focus of many indicator or valuation efforts, but a need remains for further research into quantifying the currently tenuous links between diversity and human wellbeing.

2.2 Indicators of quantity

Indicators and measures of quantity can be developed at the population, species and ecosystem level. They can express the total number or changes in number at these levels. Widely used indicators of quantity include those that highlight changes in ecosystem extent (e.g. forest area; FAO 2001) and those that demonstrate changes in species abundances (e.g. number of waterbirds; Revenga and Kura 2003). Many of these indicators focus on functional groups rather than taxonomic groupings (e.g. waterbirds, pelagic fish, wetland ecosystems).

When these measures or indicators exist for ecosystems, species or functional groups, and are coupled with good data on the benefit flows and associated economic value of those features being assessed (e.g. fish stocks (FAO 2000) or wetland services (Finlayson et al. 2005)), then these measures form a valuable indicator for demonstrating the economic impacts of biodiversity change. At the global scale changes in important fish stocks have been directly valued (e.g. Wood et al. 2005), while at local scales temporal changes in ecosystem extent have been used to quantify declines in water, erosion control, carbon storage and nature-based tourism (Reyers et al. 2009).

Indicators of quantity also include measures of primary productivity and biomass. These may be seen as undiscerning indicators of biodiversity, in that they do not measure taxonomic or functional units of species or ecosystems (but see Costanza et al. 2007). However, they are potentially useful indicators of ecosystem production which has been linked to several benefits including carbon storage (Naidoo et al. 2008), timber production (Balmford et al. 2008) and grazing (O'Farrell et al. 2007). They currently do not differentiate between natural / indigenous production and human enhanced production, and must therefore be carefully interpreted when calculating the economic consequences of ecosystem and biodiversity change for a specific area.

Data gaps include clear geographical and taxonomic selection biases towards popular, well known and easy to measure species and ecosystems e.g. mammals, birds, forest ecosystems (Royal Society 2003; Collen et al 2008; Schmeller et al. 2009). Further gaps in knowledge and data on the abundance of, for example, useful plants and animals, limit the development of these indicators and may result in a significant underestimation of the economic impacts of species and ecosystem losses.

In reviewing their ability to convey their message, strengths of indicators include their intuitiveness (especially measures of well known species and ecosystems e.g. fish and forests), the general consensus on methods and data, and their sensitivity to change. Weaknesses exist around the methods, use and communication of measures of productivity and biomass, but good progress is being made (Imhoff et al. 2004).

Due to the clear links to easily valued provisioning services, this category of measures and indicators holds much promise for measuring and predicting some of the economic consequences of change in biodiversity and ecosystem services. At a local scale this is already possible where data on ecosystem extent and species abundances exist (Balmford et al. 2002; Reyers et al. 2009); at a global scale this will require the rapid development and collation of global databases on ecosystem extent and information on the abundance of a wide range of useful species, and changes in these measures. Current ability to model changes in ecosystem extent (Czúcz et al. 2009), as well as changes in species abundances (Scholes and Biggs 2005; Diaz et al. 2006; Alkemade et al. 2009) make this a useful focus for TEEB. A focus on functional types or groups could prove highly useful and help avoid the challenges associated with the issue of redundancy, where more than one species or ecosystem is capable of providing a service (Diaz et al. 2006).

2.3 Indicators of condition

These measures reflect changes in the condition or quality of ecosystems and biodiversity, reflecting the degradation of components of biodiversity from the population level to the ecosystem. While they are closely linked to the previous category, these indicators focus less on the quantity of species or ecosystems, and more on the quality or integrity of the element being assessed. Examples include species and ecosystems at risk of extinction (Mace and Lande 1991; EEA 2009), levels of nutrients (e.g. soil condition, nitrogen deposition and depletion; MA 2005b), degree of fragmentation of an ecosystem (Rodriguez et al. 2007), trophic level changes (Pauly et al. 1998), population integrity measures (Scholes and Biggs 2005) and alteration of disturbance regimes (Carpenter et al. 2008).

Changes in species abundances in relation to thresholds (Mace and Lande 1991), and more recently changes in ecosystem extent in relation to thresholds (Rodriguez et al. 2007), have been used to develop risk assessment protocols that highlight biodiversity features with a high risk of extinction. The Red List Index, a composite measure summarizing the overall rate at which a group of species is moving towards extinction, e.g. European birds (EEA 2009), has been widely applied and used in measuring progress towards the Biodiversity 2010 Target. These approaches could prove useful in valuing the economic risk of biodiversity loss, especially if the species and ecosystems under assessment have a high risk of extinction and are clearly linked to benefits, but this has yet to be explored.

Indicators of population integrity include recently developed composite indices focused on changes in abundance. Examples of these are the Biodiversity Intactness Index (BII) (Scholes and Biggs 2005) and the Mean Species Abundance (MSA) Index (Alkemade et al. 2009). These indices use data and expert input on land cover and use impacts on populations of species, and together with information on historic or predicted land use change, model the aggregated impact of change at a population level. While useful tools for assessing the population level consequences of land use change, these mean or summed aggregated measures make it hard to link these changes to shifts in benefit flows (which are usually linked to only a few species, functional types or populations within the set modeled). The disaggregateable and traceable nature of the BII makes this a useful indicator of biodiversity condition, and with more research and data could be extended to measure functional group integrity – providing a clearer link to benefits.

MSA is an index that captures the average effect of anthropogenic drivers of change on a set of species. This measure provides insight into the effects of disturbance, particularly land cover change on species numbers, with the focus on determining the average numbers of species for disturbed versus undisturbed environments. It is also linked to various global scenarios, useful in the context of TEEB. This indicator is not an independently verifiable measure and is strongly influenced by the assumed species assemblages at the outset. Because it is a measure of the average population response, the same MSA values can result from very different situations. Furthermore, this average effect is unable to deal with changing species composition such as extinction or invasion and will miss important functional changes associated with the loss of particular species. This, together with an inability to incorporate changes in ecosystem functions resulting from biodiversity loss, makes the index's links to ecosystem services potentially tenuous.

Many of these measures and indicators have been applied at global and sub-global scales (e.g. MA 2005a; Biggs et al. 2006; EEA 2009) and appear to provide a clear and relevant message on the condition and trends of biodiversity. Some data and methodological gaps exist for determining ecosystem fragmentation or alteration of disturbance regimes and limit these indicators to use at mostly sub-global scales. Generally, these indicators are data and knowledge intensive (but can be supplemented by expert input), and are often only available at sub-global scales.

As they currently stand, few of these condition measures are amenable to the aims of TEEB, but their wide uptake, ease of application and available data and models will make them central to most assessments of biodiversity and ecosystems. However, their links to benefit provision are however tenuous and complicated by inadequate knowledge of the relationship between ecosystem integrity and benefit flow, as well as gaps in our knowledge of functional thresholds.

2.4 Indicators of pressures

In many cases the measurement and modelling of ecosystem and biodiversity change relies on measures of the pressures facing biodiversity and ecosystems as an indicator of biodiversity loss or ecosystem change. These pressures include many of the direct drivers of change highlighted by the MA as the most important factors affecting biodiversity and ecosystems: habitat destruction, introduction of alien invasive species, overexploitation, disease and climate change (Mace et al. 2005). These measures rely on land cover and use data, climate change models, distribution and density data on alien species and data on levels of use. Some indicators are composite indices which incorporate several pressures to indicate human impacts on ecosystems. Chief examples include the Living Planet Index (www.panda.org/livingplanet), the Ecological Footprint (www.ecologicalfootprint.com), and Human Appropriated Net Primary Productivity (HANPP: Erb et al. (2009), Imhoff et al. (2004) and with specific reference to biodiversity Haberl et al. (2007); for maps see http://www.uni-klu.ac.at/socec/inhalt/1191.htm) which are all available at a global scale over a period of time. Many of these composite measures also include thresholds of carrying capacity or total annual productivity to provide an indication of the sustainability of these impacts.

Land cover change is a widely used measure, where remote sensing and satellite imagery have made such data available for all parts of the world (e.g. Global Land Cover (GLC2000); Bartholomé and Belward 2005). Time series data on land cover, as well as models of future land cover change have been used to assess biodiversity and ecosystem service change at all scales from global (Mace et al. 2005) to local (Fox et al. 2005). Levels of pollutants and eutrophication are commonly used measures of human pressures at a global scale (MA 2005b; EEA 2009).

The history and widespread use of many of these pressure measures demonstrates their sound ability to convey the message of human pressures on biodiversity. The recent additions of composite indices relative to some threshold capacity have proven a useful and relevant communication tool.

While land cover should not be confused with ecosystems, these data can still be useful for broad assessments of changes in benefit flows associated with particular classes of land cover. Some of the earliest work on quantifying the economic consequences of land cover change was done in this fashion by Costanza et al. (1997). A few local-scale studies which attempt to measure change in ecosystem services rely on this approach using land cover change data (derived from remote sensing) and ecosystem service value coefficients (usually extracted from Costanza et al. 1997) (Kreuter et al. 2001, Zhao et al. 2004, Viglizzo and Frank 2006, Li et al. 2007). Case studies and simulations of land cover change have also been used to examine the effects on single ecosystem services or processes (nitrogen levels: Turner II et al. 2003; pollination: Priess et al. 2007; livestock production services: O'Farrell et al. 2007; soil organic carbon: Yadav and Malanson 2008). Recent advances in ecosystem mapping (Olson et al. 2001), earth observation (Bartholomé and Belward 2005) and valuation (Chapter 5) should make this kind of approach a complementary and practical way to evaluate the economic consequences of biodiversity and ecosystem change. The ability to use data on drivers of change to predict future change provides a further compelling reason for the adoption of this indicator (e.g. Schroter et al. 2005).

2.5 Indicators of ecosystem services

Several measures of ecosystem services are already in existence and a recent review by Layke (2009) of the indicators used in the Millennium Ecosystem Assessment and its sub-global assessments highlighted that current ecosystem service indicators are limited by insufficient data and an overall low ability to convey information. Of the indicators available, Layke (2009) found them inadequate in characterizing the diversity and complexity of the benefits provided by ecosystem services. Layke (2009) found that regulating and cultural services fare worse than provisioning services in all findings.

Provisioning services were found to have a high ability to convey information for services of food, raw materials, fuel and water provision, but data availability was average and in the case of wild food, capture fisheries and aquaculture it was poor. Genetic resources and biochemicals were found to be both poor at conveying information and poor in terms of data availability.

For the cultural services Layke (2009) found no measures of spiritual or religious values and the measures of tourism, recreation and aesthetic value available showed poor data availability and poor ability to convey information. Balmford et al. (2008) highlight this shortcoming and point to a need to focus on cultural services which better lend themselves to measurement and assessment. They suggest a focus on services like bird watching and scuba diving, where the links between biodiversity and the cultural or recreational benefit are simple and clearly defined, and where valuation studies already exist (e.g. Losey and Vaughan 2006; Tapuswan and Asafu-Adiaye 2008; Lee et al. 2009). Protected area visitor numbers and values are also a potential indicator. However, these indicators are not yet available at global or regional scales.

For regulating services measures are limited to just more than half of the ecosystem services listed by Layke (2009) and where measures do exist data availability and ability to convey information are poor. Water regulation and water purification are listed as the only measures with a high ability to convey information, but are limited by data availability, while climate, air quality and natural hazard regulation were all found to have an average ability to convey information but were also hampered by an average (in the case of climate) to poor data availability.

These shortcomings in all services, but particularly in cultural and regulating services will have serious consequences for the comprehensive economic valuation of all ecosystem services, limiting the valuation to a few provisioning and even fewer regulating services.

However, the review presented in Balmford et al. (2008) as well as some recent studies (e.g. Troy and Wilson 2006, Naidoo et al. 2008; Wendland et al. 2009), indicate that the spatially explicit measurement of ecosystem services at regional and global scales is a rapidly growing research area. Projects such as those of the Heinz Center in the USA (The Heinz Center 1999; Clark et al. 2002; The Heinz Center 2006) and the European-based Advanced Terrestrial Ecosystem Analysis and Modelling (ATEAM) have made good progress in the development of indicators and the mapping of ecosystem services, even to the point of including scenarios of future change (Metzger et al. 2006).

Furthermore, the development of several international programs advancing the measurement and valuation of ecosystem services will help to fill these gaps in the future (e.g. The Natural Capital Project (Nelson et al. 2009), The Global Earth Observation Biodiversity Observation Network GEOBON (Scholes et al. 2008); The World Resources Institute Mainstreaming Ecosystem Services Initiative http://www.wri.org/project/mainstreaming-ecosystem-services/tools). These programs are developing tools and approaches to model, map and value the production of particular ecosystem services based on abiotic, biotic (often from measures listed above) and anthropogenic factors, as well as knowledge of relationships between these factors (Figure 1 presents an example from South Africa where data on rainfall, geology (lithology), vegetation type, recharge, groundwater-quality (electrical conductivity) and land use activities were used to map water flows). Very complex measures relying on species diversity, abundance, distribution and landscape pattern have also been developed at local scales (e.g. pollination; Kremen 2005).

However, Naidoo et al. (2008) observed that evidence of the spatial estimation of ecosystem services and the flow of benefits to near and distant human populations is limited to a few local case studies. Most of the existing quantitative analyses still tend to provide aggregated values for large regions, and data availability and disaggregation of spatial data are still a limitation to the mapping of ecosystem services. Furthermore, the multivariate nature of these ecosystem service indicators makes it hard to isolate the role of biodiversity in ecosystem service supply, which in turn makes the economic consequences of biodiversity loss hard to untangle from the other biotic, abiotic and anthropogenic factors involved in service supply. These are further explored in the following sections.

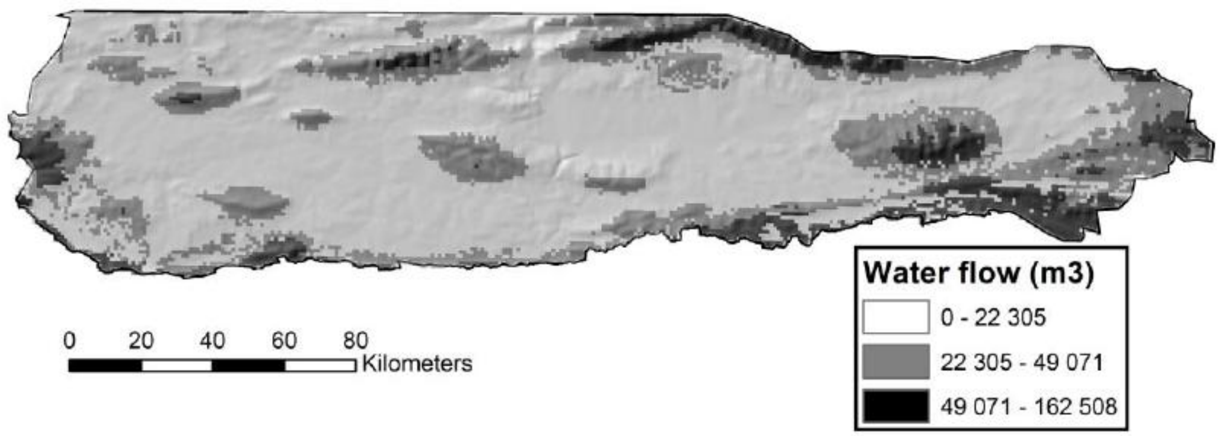

Figure 1: **Map of ecosystem services of water flows for the Little Karoo region of South Africa**. These data were used by Reyers et al. (2009) to assess changes in ecosystem services supply over time.

In summary, few indicators at present move beyond the quantification of a stock or flow of a service to the actual valuing of the service, and despite developments, calculating the contribution of biodiversity and effects of changes in its state to these values remains a challenge.

2.6 Lessons

Despite the array of biodiversity and ecosystem service indicators available, few lend themselves to a direct application of determining the economic consequences of biodiversity and ecosystem change. We will need a representative set of indicators to ensure that all relevant aspects of biodiversity and ecosystem change are captured and valued – from diversity to condition. A reliance on existing indicators will in all likelihood capture the value of a few species and ecosystems relevant to food and fibre production, and will miss out the role of biodiversity and ecosystems in supporting the full range of ecosystem services, as well as their resilience into the future.

An alternative avenue is to focus on pressures and their use in models of the economic consequences of policy inaction in the arena of land cover or climate change. This approach bypasses the actual measurement or modelling of biodiversity and ecosystem change, and investigates the implications for land cover and climate change on ecosystem services directly (e.g. Schroter et al. 2005; Metzger et al. 2006). However, this approach will not necessarily advance the case for biodiversity and ecosystem governance which is the key purpose of TEEB, but it will perhaps highlight the need for land use and climate policy and action in the context of ecosystem service governance.

While it is important to use available tools to meet short term policy and decision maker needs, it is critical to marry these measures of quantity and drivers, with measures of diversity and condition in order to ensure a full accounting of the value of biodiversity and ecosystems into decision making. So the current focus on synthesising existing data must be complemented with active research and development into the measurement of biodiversity and ecosystem change, their links to benefit flows and the value of these flows.

Many of the measures currently available are primarily determined by the existing information, which does not necessarily make them good measures or good indicators. TEEB and other assessments of the economic value and consequences of biodiversity loss will need fit-for-purpose indicators and new data with which to populate them. These fit-for-purpose indicators must address the challenges outlined in Section 3.1 and must not only be relevant and effective in conveying their message, but must also be precise, applicable across relevant systems and places, repeatable and defensible, and demonstrate a clear link between the benefit and the component of biodiversity delivering that benefit.

3 In search of relevant indicators for ecosystem services

3.1 Developing relevant indicators

It is clear from section 3.2 that most existing measures of biodiversity, ecosystems and ecosystem services were not developed for the purpose of TEEB and similar projects: to examine the economic consequences of changes in biodiversity and ecosystem services, and in particular the marginal loss of biodiversity. The InVEST model of the Natural Capital Project does allow for the quantification of economic values and changes in these values under future scenarios and is a powerful tool being explored by many global and sub-global programs (Daily et al. 2009). Rather than argue for a single unified methodology that can apply to all possible circumstances, several parallel approaches and ways of modeling are needed. To support the development of indicators relevant to the aims of TEEB and other projects interested in the economic consequences of biodiversity loss, a few potential indicators are explored below, using them to highlight key opportunities and constraints in these indicators. These include a readily measured provisioning service of timber production, a published model and map of the regulating service of carbon sequestration, and a preliminary assessment of the methods for measuring a less easily measured cultural service of social value of agricultural landscapes. The section also discusses advances made at local scales in indicator development, and ends with some discussion on the importance of baselines and thresholds in indicator development.

3.2 A provisioning service: timber production

Provisioning services (with clear production functions) appear to have received most of the attention in ecosystem service mapping exercises (Balmford et al 2008; Naidoo et al. 2008). A popular provisioning service, timber production, can be modeled and mapped using estimates of DMP - Dry Matter Productivity in forest areas by combining remote sensing imagery with meteorological data (for more information see http://geofront.vgt.vito.be/geosuccess/relay.do?dispatch=DMP_info). The service's production function includes measures of ecosystem extent (forest area) and measures of biological quantity (dry matter). The DMP index provides a measure of the vegetation growth in kilograms of dry matter per hectare. This is the annual amount of new dry matter created by the ecosystems and can be understood as the new timber offered by the ecosystems each year. Comparing DMP across different years can show areas with different vegetation activity, enabling it to be used to derive changes in DMP and find those areas where natural timber production has increased or decreased.

The maps in Figure 2 show a world map which illustrates where the Dry Matter Productivity is more intense (darker green colour). The country-scale maps show the difference between the years 2001 and 2004 for the vegetation activity in West Africa and in Madagascar. In the case of Madagascar, the total DMP dropped by 6% between 2001 and 2004 due to deforestation.

This measure provides good opportunities to measure and model the impacts of changes in forest area and natural timber production (measures of the quantity – Table 1) on the production service. However, it still falls short of the ideal TEEB indicators in that some development is still required in converting DMP units into economic value (i.e. determining commercially important species, use and access). Further useful development could also include information on levels of sustainable production by using, for example the weight of dry matter per hectare grown in a specific year. Disentangling the role of biodiversity from the anthropogenic factors associated with timber production will be a challenge.

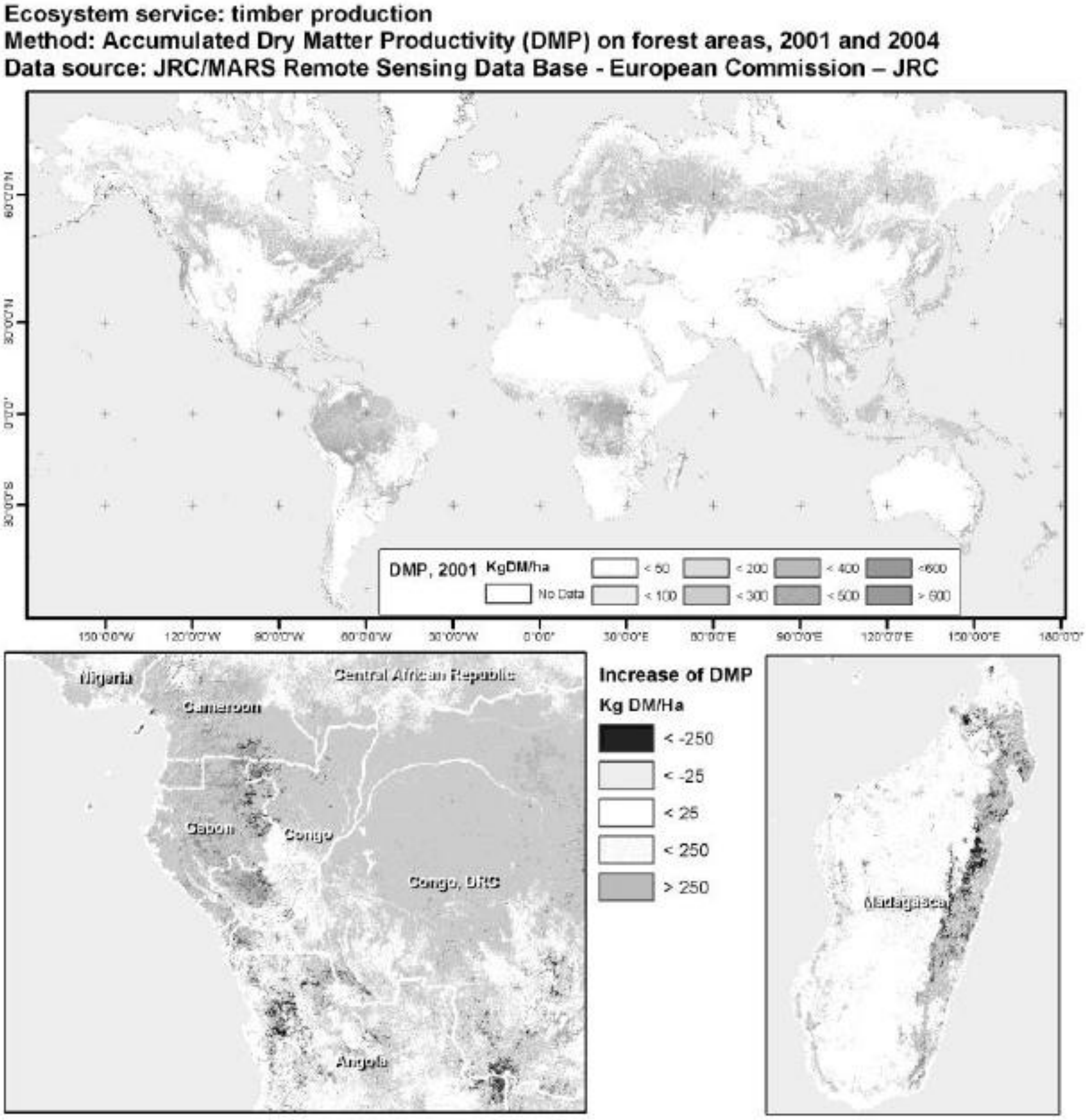

Figure 2: **Maps of timber production, measured as Dry Matter Productivity (DMP) in forest areas for (a) the world in 2001, and highlighting change in timber production between 2001 and 2004 for (b) West Africa and (c) Madagascar**.

Source: JRC/MARS remote sensing data base – European Commission – JRC

Forests provide a bundle of ecosystem services including carbon sequestration, scenic values, watershed protection and cultural services. These services interact with one another in a dependent and non-linear fashion. Harvesting timber will cause declines in many other services from forest (which are more challenging to measure). Quantifying and managing these trade offs is a key challenge to sustainable development. By taking a single service approach – like the timber production service described here – the other services and their values are ignored.

This highlights the importance of taking a multi-service approach to economic valuation taking account of trade-offs over ecosystem services, space and time.

3.3 A regulating service: global carbon sequestration

Ecosystems play an important role in determining atmospheric chemistry, acting as both sources and sinks for many atmospheric constituents that affect air quality or that affect climate by changing radiative forcing. This ability of ecosystems to modify the climate forms the ecosystem services of climate regulation. Carbon sequestration, the removal of carbon from the atmosphere by the living phytomass of ecosystems, is an important component of this climate regulation service. In the map below (Figure 3) carbon sequestration was modeled as the net annual rate of atmospheric carbon added to existing biomass carbon pools, using a proxy of net carbon exchange (NCE) produced in simulations using the Terrestrial Ecosystem Model (TEM) developed by McGuire et al. (2001) and applied by Naidoo et al. (2008). The model simulates carbon exchange between the atmosphere and terrestrial biosphere on the basis of vegetation types, soils, climate, atmospheric CO2, and land use history.

Naidoo et al. (2008) point to the limitations of using a model based rather than observational approach and the reliance on assumptions, time series and input variables. However, together with the possibility of assigning economic values to the tons of carbon sequestered, Balmford et al. (2008) point to the possibilities presented by these land-use-coupled models to estimate differences in carbon storage, emissions and sequestration under different scenarios (e.g. McGuire et al. 2001). This would make it possible to map the economic value of these services of global climate regulation, and how they might change under different scenarios of ecosystem and land use change.

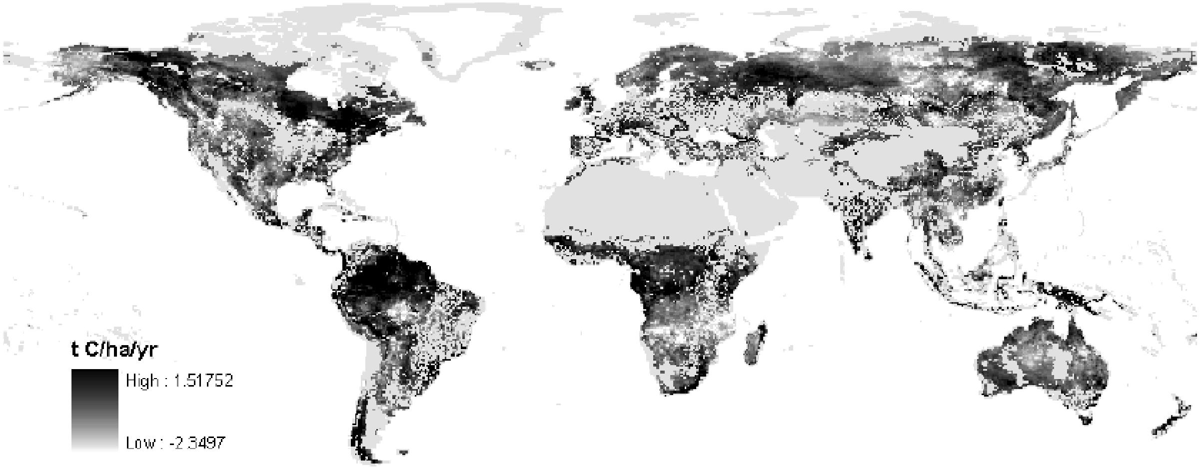

Figure 3: Global map of carbon sequestration developed by Naidoo et al. (2008) using the Terrestrial Ecosystem Model (TEM) developed by McGuire et al. (2001). Source: National Academy of Sciences, U.S.A, 2008.

3.4 A cultural service: social appreciation of agricultural landscape

Cultural ecosystem services refer to the aesthetic, spiritual, recreational, educational and other non-material benefits that humans obtain from contact with ecosystems (MA 2005b; Butler and Oluoch-Kosura, 2006). Little progress has been made in mapping cultural services. Even in the case of the popular cultural service of nature-related outdoor tourism, Balmford et al. (2008) point out that these services or their benefits cannot yet be mapped due to both a lack of knowledge on the links between biodiversity and tourism demand or use and the subjective and context-specific nature of perception and appreciation. Nevertheless attempts can be made to quantify and map cultural services on the basis of proxies that describe societal interest for cultural ecosystem services in specific landscape types. This example represents an attempt to derive an index of social value of the agricultural landscape.

It is currently not possible, in the context of a global or regional assessment, to address landscape perception through targeted enquiries and the use of questionnaires to record people’s preferences. Instead in this example three variables were identified as representative of societies’ preferences: protected agricultural sites; rural tourism; and presence of labeled products and combined in the map shown in Figure 4.

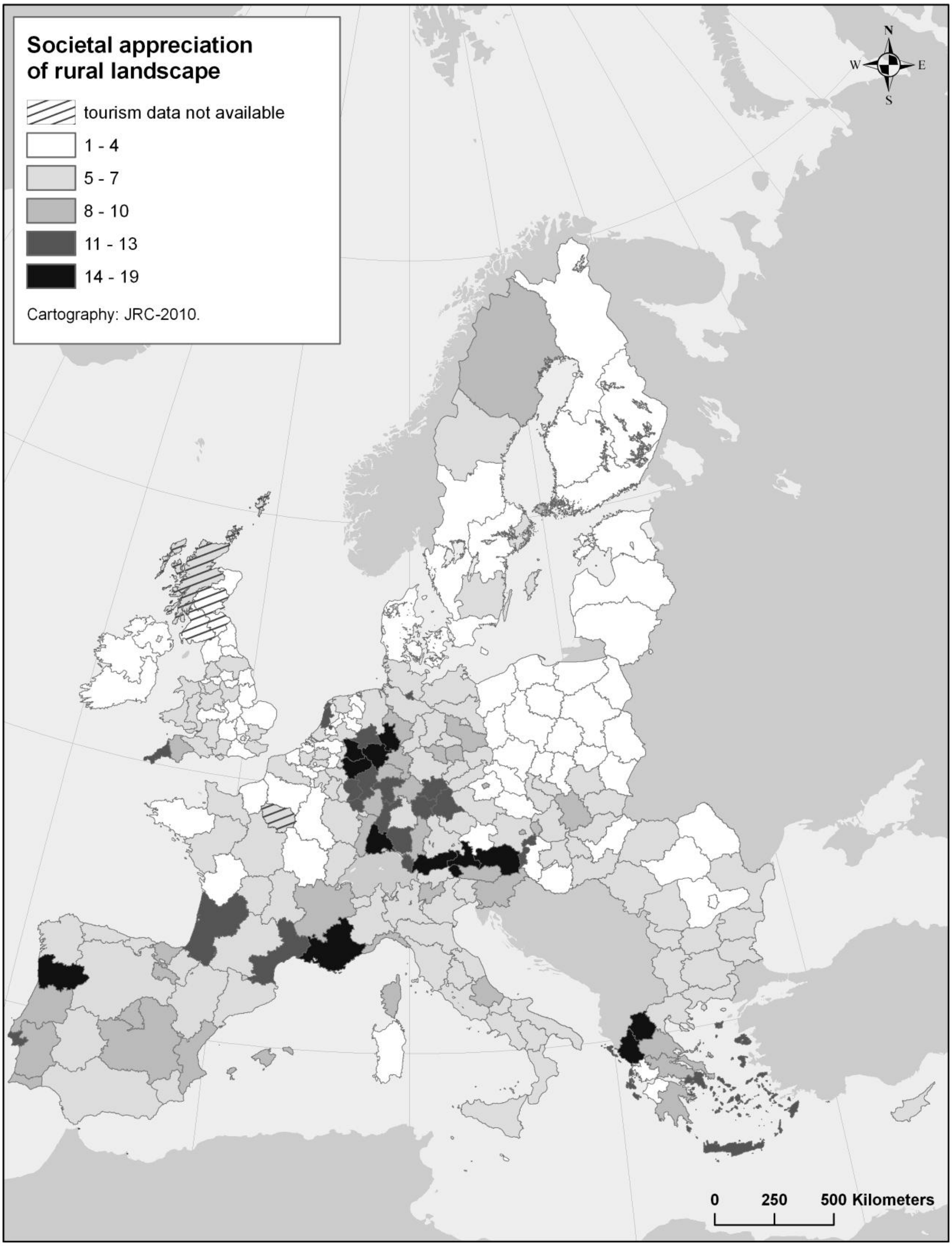

Figure 4: **Social value of agricultural landscapes in Europe determined by protected agricultural sites; rural tourism; and presence of labeled products**

This example of a spatially explicit cultural service, although a novel demonstration of the distribution of regional social value, is still some distance from a fit-for-purpose indicator which can be used to measure and model the economic consequences of biodiversity and ecosystem change. Methods for mapping cultural services are not yet developed or agreed and therefore this model has to be carefully interpreted, in order to avoid the risk of confounding different values or assuming direct transfer of values. Trade-offs and synergies in the input components must be understood and correctly taken into consideration, and underlying measures would have to be made available in a format that can be disaggregated and traceable. Furthermore, assigning economic values to social values will be a challenge (see Chapter 5 for developments in this area), while determining the changing contribution of biodiversity and determining past and future trends in the service are also not yet possible. As Balmford et al. (2008) suggest it might be best to start with cultural services where the links between biodiversity and the cultural or recreational benefit are simple and clearly defined and where valuation studies already exist (e.g. bird watching; Lee et al. 2009). Protected area visitor numbers and values are also a potential indicator which should be explored at regional and global scales.

3.5 Relevant indicators at local scales

The above global and regional scale indicators highlight some of the opportunities and challenges faced in the search for indicators of the economic consequences of changes in biodiversity and ecosystems, and in particular indicators that are convertible into economic values. At the local scale recent publications highlight the progress made in ecosystem service indicator development. Chan et al. (2006), Nelson et al. (2009) and Reyers et al. (2009), used data from a variety of sources on ecosystems and biodiversity (especially functional types), land cover, population, access, hydrology and economic value to model and map multiple ecosystem services at a local scale in the USA and South Africa. These maps were used to investigate trade-offs and planning options by Chan et al. (2006), to quantify the consequences of land use change on ecosystem services by Reyers et al. (2009) and to investigate the consequences of future scenarios on ecosystem services by Nelson et al. (2009).

While some of the indicators are expressed in biophysical quantities, these quantities (litres of water, tons of carbon) are convertible into economic terms. This conversion is clearly demonstrated in another local scale study by Naidoo and Ricketts (2006) in Paraguay where the value of ecosystem services was modelled and made spatially explicit to assess the costs and benefits of biodiversity conservation in the region.

3.6 The way forward

This review has highlighted the following lessons for mapping ecosystem services for economic valuation and for use in scenarios of the marginal costs and benefits of ecosystem change and biodiversity loss:

- The need to be spatially explicit while not resorting to large regional aggregations reduces the set of ecosystem services which currently can be mapped to mostly provisioning services at global and regional scales
- Global datasets of primary productivity and vegetation cover have played a significant role in most of the global maps of services now available (e.g. Naidoo et al. 2008)
- Ecosystem service mapping needs to progress beyond the production of maps that show biophysical quantities or biological stocks of services such as grazing resources to an approach that includes regulating and cultural services and the relationships between these services (i.e. an approach that is cognizant of trade-offs)
- Spatially explicit data on the flow of services and their use at global scales are rare, proving a major obstacle to moving from maps of biophysical quantities to maps of economic value. This is less of an issue at local and regional scales.
- Few of the existing global, regional and local maps of ecosystem services demonstrate clear and indisputable connections between biodiversity to the final benefit quantity or value
- Investment in spatially explicit data and local and regional scales are a first necessary step in improving ecosystem service mapping and in turn economic valuation
- Alignment between available maps of ecosystem services and existing models or scenarios of future change is limited, making it difficult for the assessment of change in service levels and values.

Finally a lesson emerging from this chapter is that using global maps of service production, and changes in these services, as a proxy of value and value change, may miss out on two crucial facets related to ecosystem management thresholds: sustainability and vulnerability. The challenge of sustainability can be highlighted in the case of fisheries where Figure 5 shows fish stocks outside safe biological limits (from http://themes.eea.europa.eu/IMS/ISpecs/ISpecification20041007132227/IAssessment1199788347728/view_content) which would not necessarily be captured by a map of trophic biomass.

This highlights the crucial importance of thresholds in ecosystem service measures and indicators.

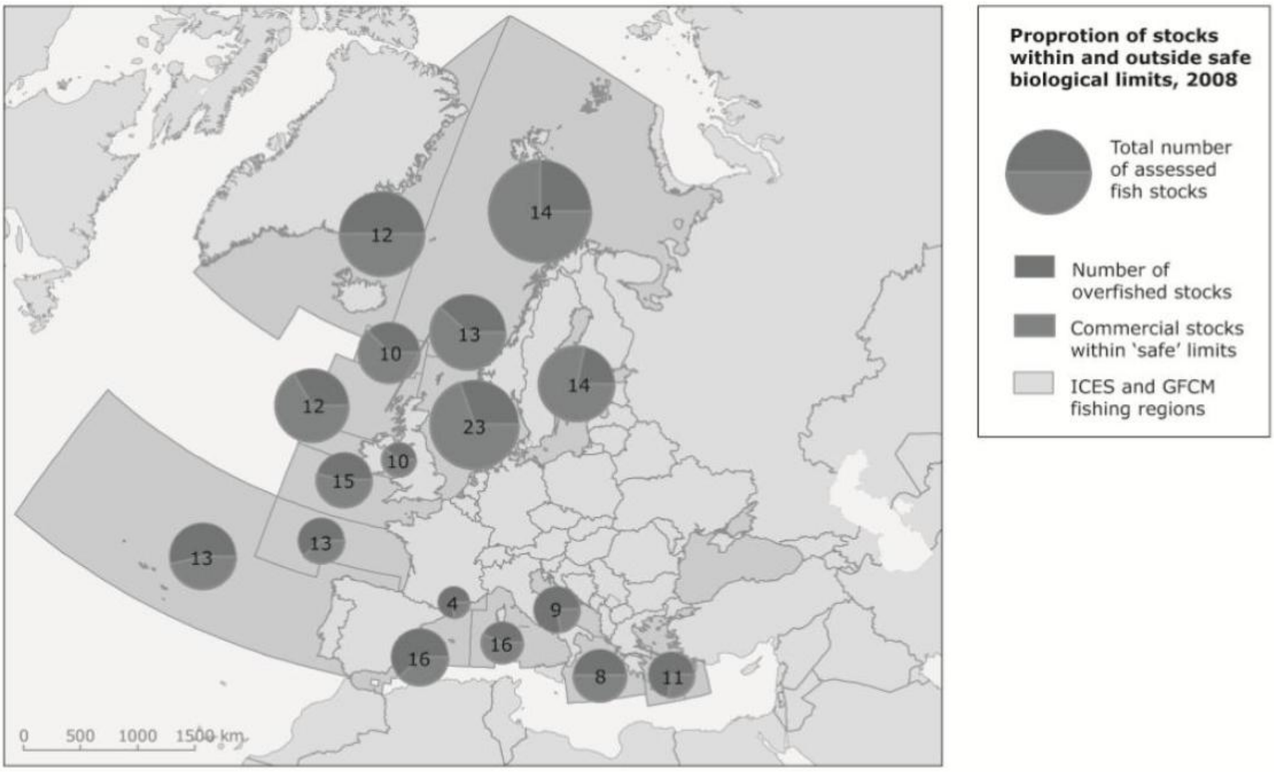

Figure 5: **Fish stocks outside safe biological limits,** extracted from http://themes.eea.europa.eu/IMS/ISpecs/ISpecification20041007132227/IAssessment1199788344728/view_content). The chart shows the proportion of assessed stocks which are overfished (red) and stocks within safe biological limits (blue). Number in circle is the number of stocks assessed within the given region. The size of the circles is scaled proportional to the magnitude of the regional catch.

In demonstrating the challenge of depicting vulnerability, Figure 6 shows a map of Southern Africa where provision of water has been displayed as a proportion of demand for water (Scholes and Biggs 2004). This map highlights areas of high vulnerability where water supplies do not currently meet water demand. In global or regional maps of this service, high value areas do not accurately depict important areas with a low water supply where social thresholds and local demand are not met. Even small changes in water supply in this important and vulnerable areas would have significant impacts on human wellbeing in those areas, impacts that would not necessarily be illustrated in an assessment of monetary value and changes in that value. Closer examination and differentiation of the demand for services, which should theoretically be linked with supply, may provide a more socially realistic assessment of services.

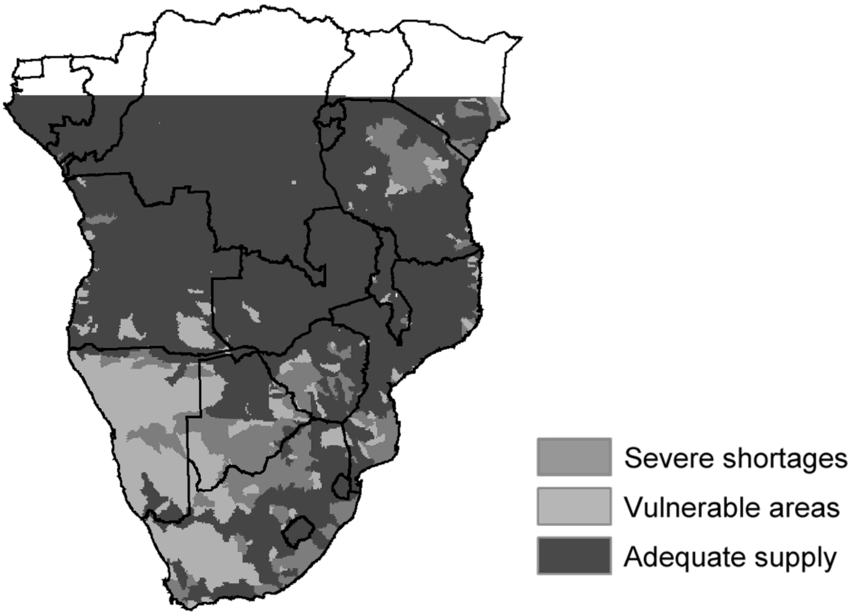

Figure 6: Water availability in Southern Africa expressed as relative to demand for water

Source: Scholes and Biggs 2004

Both of these examples reflect the importance of thresholds highlighted in the MA. Ecosystem service change is seldom linear or independent and can often be accelerating, abrupt and potentially irreversible (MA 2005b). The loss of biodiversity and increasing pressures from drivers of ecosystem change increase the likelihood of these non-linear changes. While science is increasingly able to predict some of these risks and non-linearities, predicting the thresholds at which these changes will happen generally is not possible. Users of indicators and assessments of ecosystem change and its consequences need to bear this in mind, and where possible to reflect known or possible ecological and social thresholds and not to assume linear relationships between biodiversity loss and its consequences.

4 Link to valuation and further work

The flow of ecosystem services from point of production to point of use is influenced by both biophysical (e.g. currents, migration) and anthropogenic (e.g. trade, access) processes which influence the scale of service flow from locally produced and used services (e.g. soil production) to globally distributed benefits (e.g. carbon sequestration for climate regulation). The flow of benefits and scale of flows influences the value of the service due to changes in demand and supply which vary spatially and temporally. Use of the service is intrinsically a human centered process relying largely on socio-economic data and to a lesser degree on biophysical information. Information will include the distribution of users, the socio-economic circumstances of users, governance systems, human pressure on ecosystems and other social measures like willingness and perceptions. Spatial data are likely to include maps of population distribution and economic status, maps of land use, trade data and other spatial data on political units and administrative boundaries.

In order to make a comprehensive and compelling economic case for the conservation of ecosystems and biodiversity it is essential to be able to understand, quantify and map the benefits received from ecosystems and biodiversity, and assign values to those benefits. This all must be done in a fashion that makes it possible to assess the contribution made by biodiversity to this value (separately from the contribution made by abiotic and anthropogenic factors), as well as the consequences of changes in ecosystems and biodiversity for these values. This chapter has focused on reviewing current ability to quantify and make spatially explicit the biophysical quantities (water, food, timber) or benefits provided by ecosystems and biodiversity. It has also aimed to review current ability to make spatially explicit the other beneficial processes from ecosystems and biodiversity which form our life support systems (e.g. pollination, carbon sequestration and cultural services).

Chapter 5 describes in detail the methodologies used and challenges faced when attempting the valuation of biodiversity and ecosystem services. In economic valuation, the focus has been on flows of ecosystem services (the "interest" from the capital stock). Chapter 5 acknowledges that the valuation literature currently does not consider biodiversity in detail. It is indeed not straightforward to assign a value to the actual diversity in a system, as opposed to the biomass present. At the same time, the diversity is linked to production, so measurements of this aspect need to be built in. The instruments described in this chapter, on the other hand, have traditionally focused on biological resources, the capital itself. Biophysical measurements are important since biodiversity underpins the delivery of many ecosystem services and thus forms the underlying basis of value. The framework of Total Economic Value (TEV) (see Figure 2 of chapter 5 - Value types within the TEV approach) is useful to help analyse where indicators need to be further developed. Ecosystem accounting is also addressed in Chapter 3 of the TEEB D1 Report, where several elements of the accounting framework (e.g. data issues, valuation approaches, socio-ecological accounting units) are examined for the three interconnected governance levels, Global/Continental, National/Regional, and Local.

[1] The data availability and ability to convey messages for ecosystem service measures and indicators are reviewed in detail by Layke 2009

Chapter 3

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Chapter 4

Socio-cultural context of ecosystem and biodiversity valuation

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Contents

Key messages

- Valuation, including economic valuation, functions as a system of cultural projection which imposes a way of thinking and a form of relationship with the environment and reflects particular perceived realities, worldviews mind sets and belief systems. However, it can also serve as a tool for self-reflection and feedback mechanism which helps people rethink their relations to the natural environment and increase knowledge about the consequences of consumption, choices and behavior.

- Because of the multidimensional and socio-cultural embeddedness of value any exercise of valuation is relative to a given individual or group of people. In a multi-cultural and democratic context of biodiversity valuation, this makes the question of choosing a value articulating institution more important than that of finding a correct value.

- Economic valuation influences the notion of ownership and property applied to biodiversity and over the long term may change human relationship to the environment in significant ways.

- Intrinsic values are culturally embedded moral truths. They can be taken into account by choosing the appropriate institutions which allow their articulation in addition to utilitarian values.

- Valuation processes can be seen as a form of regulatory adaptation by serving as a mechanism to provide feedback in a system where production and consumption, trade and exchange are so distant and complex that they undermine perceptions of the impacts of habits and behavior on the environment.

- Value change along the commodity chain has implications for the distribution of benefits, affects the level of incentives for conservation and represents an important methodological challenge for economic valuation.

- Economic valuation may contribute to address our inability, reluctance or ideological intolerance to adjust institutions (also those which are value articulating) to our knowledge of ecosystems, biodiversity and the human being.

- Economic valuation is a complex, spatial and institutional cross-scale problem. Many efforts focusing on particular parts of ecosystems or species, while effective at one level, lack the scope to control the pressure of commodity markets for land resources surrounding them. As such, and depending on their biophysical context, they may be limited to capturing the linkages and vertical interplay created by a growing functional interdependency of resource use systems nested within larger ecosystems.

1 Introduction

Economic valuation of ecosystems, their services and biodiversity represents a balancing act. On the one hand, valuation can function as a system of cultural projection which imposes a way of thinking and a form of relationship with the environment[i], a particular notion of property and ownership, and view of development and what constitutes human wellbeing. Side by side with contributions from several distinguished scholars such as Arrow, Sen's liberal paradox has shown the inadequacy of this worldview (Sen 1973; Arrow 1982). On the other hand, valuation can serve as a tool for self-reflection which helps people rethink their relations to the natural environment and increase knowledge about the consequences of consumption choices and behavior for distant places and people. Questions such as what factors influence human relationships with nature, what is the role played by nature in the formation of social and personal identity and what are the social and environmental consequences of various ways of relating to and using the environment (Zavestowski, 2004; Clayton and Optorow 2004) become the focus of this self reflection.

As such, valuation can work as a feedback mechanism for a society that derives its resources but has distanced itself from the environment for the consequences of its actions. Further, economic valuation can help communicate the value of nature to different people using a language which speaks to dominant economic and political views around the world. The outcomes of any valuation exercise depend on what the various interest groups value, whose values count, who benefits, and how we account for the growing interdependencies of social and ecological systems. In that context the question of how to value ecosystem services and biodiversity and the choice of a valuation method are as challenging as the attempt to attach a particular value to them. Valuation entails conceptual and methodological challenges to account not only for different dimensions of value and their interconnections, but assimilating different cultural perspectives and levels of analysis.

Across disciplines, scholars have recognized cultural differences as very fundamental in the way people conceive and relate to the environment. While it is not the intention of this chapter to compare and review these attempts, a brief overview may offer insights that are particularly relevant to the exercise of environmental valuation. For instance, Descola (1996) proposes a three tier analytical model to characterize implicit schemes of praxis [practical and applied knowledge] used by different societies to objectify their relationship to nature: 1. modes of identification including animism (i.e. social character of relations between humans and non-humans thus the space between nature and society as being social), totemism (metaphoric conceptualization of social distinctions based on the relationship between nature and culture) , and naturalism (the space between nature and society as being natural and biological); 2. modes of interaction (i.e., based on reciprocity, predation, protection), and modes of identification (i.e., metaphoric similarities like in totemic structures, and metonymic as exemplified differently by animistic and naturalistic inter-associations between species). His approach helps to place the western perspective (i.e., scientific, naturalist, and protectionist) as rooted within not only a Judeo-Christian tradition of control and utilitarianism, but also in the context of the historical separation of social and biological sciences and the consequent epistemological distinction between culture and nature. This contrasts with indigenous understandings of the human-nature relationship that acknowledge a continuum between human and non-human as part of a large chain of reciprocity and predation. In contrast, for instance, the conservation movement, as Descola puts it, "fetishing nature as a transcendental object, the control of which would be displaced from predatory capitalism to the rational management of modern economics, the conservationist movement, far from questioning the foundation of Western cosmology, tends rather to perpetuate the ontological dualism [culture-nature] typical of modern ideology." (1996: 97). Descola goes further, however, predicting that in the long run this agenda may shift completely the relationship of people to nature:

> "However, the program set forth by environmental activists will perhaps lead, unintentionally, to dissolution of naturalism, since the survival of a whole range of non-humans, now increasingly protected from anthropic damage, will shortly depend almost exclusively upon social conventions and human action. The conditions of existence for blue whales, the ozone layer or the Antarctic will thus be no more 'natural' than they are presently for wild species in zoos or for genes in biological data banks. Drifting away from its time-honored definition, nature is less and less the product of an autonomous principle of development; it's foreseeable demise, as a concept, will probably close a long chapter of our own history." (1996: 97-98).

Along similar lines, Palsson (1996) distinguishes three kinds of paradigms representing particular forms of human-environment interaction: orientalism; paternalism; and communalism. While the former two are based on different degrees of separating nature and society, the latter "rejects the radical separation of nature and society, object and subject, emphasizing a notion of dialogue." (1996: 65). Ellen (1996), on the other hand, proposes a model to interpret cultural variations on the relationship between people and nature based on a comparative perspective to human cognitive imperatives: how people identify things based on senses, context, and value; which code systems people use to contrast self and others; and, the different ways people recognize some inner force and essence in nature. However, calling attention to the danger of 'infinite relativity', Ellen's model seeks to understand the implications of different systems of thinking to our current environmental challenges. In other words (paraphrasing Ellen 1996: 28), how particular ways of conceiving the environment serve the interest of particular groups, whether these are the conservation movement, industries, churches, political parties, academics, indigenous people, or governments. Whether reinventing an image of the noble ecological savage, or defining nature with a price tag, or constructing a shared image of the global environment, these models carry political and social goals and have long far-reaching consequences (Ellen 1996: 28). An important message for TEEB studies is that approaches to understand nature and environment originally driven by naturalism have drifted away to utilitaria-led paradigm of economics as a social choice. But the limitations of utilitarian approach is not fully understood by societies, after all an utilitarian approach is essentially individualistic, from which there is no way to deduce a social welfare unless very strict conditions are laid (Arrow 1963; Sen 1970a).

Yet, while some scholars tend to focus on the contexts of people-nature relationship, others have proposed that our modes of relation to nature are also innate to human evolutionary history (Wilson 1984; Kellert and Wilson 1993). The biophilia hypothesis, for instance, articulates that all humans have an innate need and connection to nature, including spiritual and emotional, as well as utilitarian dimensions. These relationships, however, are not ahistorical and change in level of importance as a function of the different degrees of interaction and dependency on it. Kellert (1996) presents nine dimensions intrinsic to the way people and environment relate, some of which can dominate depending on our experience and context, the balance of which influence well-being: utilitarian; naturalistic; scientific/ecologic; aesthetic; symbolic; humanistic; moralistic; dominionistic; and negativistic (1996: 38). As in other domains of science and philosophy, dichotomies and typologies defining how people vary in the way they value and relate to nature can be useful or misleading, not necessarily true or false. Following the approach suggested by Ellen, the intention here in recognizing cultural differences and perspectives to nature is to find synergies between them, the utilitarian and aesthetics, the pragmatics and the symbolic (Ellen 1996).

The epistemological tradition of economics has also evolved from a specific socio-cultural context which presumes that economic values are predefined, held by people as preferences and can be derived by analysis, either via stated or revealed preference methods. Economics is built on this "belief system" as manifested in the Cartesian and Newtonian tradition of modern science, and Descola's referred naturalism, according to which the world works mechanistically as a giant clockwork. Understanding the complexity of the world requires taking it apart to its individual components or according to the Cartesian worldview, even if implying separating the mind from the body, culture from nature, so the latter can be understood objectively, free of values. In this sense, one can see this particular context, and the pre-analytic assumptions of how ecosystems function and humans behave, as part of TEEB's constructed reality and values, nevertheless based on perceived social demands and scientific methods. Gowdy (1998: xvi) expressed this by stating that "My own particular tribe, that of academic economists has its own belief system to explain and justify the world of commerce we have created, typified by the notion of "economic man.". Within the Western socio-cultural context "economic man" stems from the Judeo-Christian conception of human nature: because of Adam's original sin, inherited by humanity, humans are viewed as "scarcity-driven creature(s) of need" and "as ever imperfect and suffering beings with wants ever beyond their powers" (Sahlins 1996: 397). Others have argued that "In Hebrew religion, the ancient bond between God and nature was destroyed" (Frankfort 1948: 343), and that "Christianity continued to widen the rift between man and nature by its opposition to pantheism" (the belief that nature/the universe is a manifestation of god or spirits), thereby desacralizing nature and reframing it as natural resource) (Gatzweiler 2003: 61).

As such, economic values and processes of valuation, although grounded in a shared scientific methodology, are socially and culturally constructed, as are concepts such as ecosystems and biodiversity. Economic values are not objective facts nor do they reflect universal truths; instead they reflect the culturally constructed realities, worldviews, mind sets and belief systems of particular societies and/or sectors of society (Wilk and Cligget 2006). They are not exogenous, but rather shaped by the social interactions of everyday life (Henrich et al. 2001) as well as political and power relations operating within a system of local, regional, and global interdependencies (Hornborg et al 2007). They derive from a belief system of how economists and others view the world and what they think the role of humans in it should be. Mainstream economic beliefs of values of ecosystems and biodiversity are defined by people's willingness to pay for them and the existence or creation of markets, but they have to be understood as part of the broader historical and geopolitical context which gave rise to contemporary environmental conservation and valuation.

The rise of the so-called new-environmentalism during the 1960s marked a shift from environmental concerns based on the protection of "empty" spaces and particular species to concerns with the human environment (McCormick 1989; Caldwell 1990). This is well represented by the 1972 United Nations Stockholm conference on 'The Human Environment'. Since then, although many efforts have aimed at regulating development and reconciling economic growth and conservation, the most striking outcome of these processes has been the exponential rise of protected areas (Zimmerer 2006). Some have referred to this process as the globalization of conservation for its generalized world distribution and impact on local populations (West et al. 2006). Even today, more political emphasis is placed on protecting and isolating ecosystems from economic development and commodity markets, than on redefining and regulating the latter.

During the 1980s, the now landmark Brundtland Commission of the United Nations 'Our Common Future' report (Brundtland 1987) marked more directly an effort to internalize the environment in the economy, slowly opening space for new conceptions of development based on the principle of intergenerational responsibility. The latest Global Environmental Outlook (GEO 4) of the United Nations Environment Program (UNEP), for instance, which has started as a periodic assessment exercise since 'Our Common Future', positively acknowledges the mainstreaming of environmental issues in government and corporate agendas during the past 20 years, i.e., "from periphery to the core of decision making" (UNEP 2007; King et al. 2007).

Reflecting these larger trends in conservation, the Convention of Biological Diversity (CBD), one of the lasting high-profile agreements resulting from the UN Rio 1992 Earth Summit, represented a shift from species-based conservation, championed by some international conservation organizations, to the conservation of ecosystems and biomes. It also gave heightened importance to local populations as stewards of nature and as a source of knowledge relevant to conservation and sustainable development. Not unlike TEEB, it has put significant emphasis on the value, including economic value, of biodiversity and local knowledge, such as through incentives for bioprospecting (Reid et al.

1993), a process which triggered diverse social consequences, many of which are still unfolding (Moran et al. 2001).

The Millennium Ecosystem Assessment (MA) represented another important shift to the efforts to view the environment at a global scale and internalize it in policy and economic thinking. Its core concept of ecosystem services, for instance, while emphasizing an anthropocentric and utilitarian approach, proposes a framework centered on human dependency not only on resources, but on ecosystem functioning itself, contributing to make visible a broad array of ecological and biophysical functions taken for granted by society (MA 2005). In doing so, the MA has contributed to a broader understanding of the overwhelming scale of human impacts and their footprints, and their current and future economic and social consequences.

Valuation exercises have not shied away from the challenges of valuing biodiversity and ecosystems from different social and cultural perspectives, and levels of analysis. Many recognize the multiple dimensions and concepts of value embedded in ecosystems and biodiversity, and that any exercise of valuation is relative to a given individual or group of people (Turner et al. 2003; Shmelev 2008). A comprehensive report from the United States Environmental Protection Agency (EPA), for instance, proposes an approach of multiple methods to help capture different dimensions of biodiversity and ecosystems services, as well as the perspectives of different stakeholders. The underlying idea is that an integrated and multi-dimensional approach will be more likely to capture the full range of contributions, thus the broader value, of biodiversity and ecosystems, including values which may be context specific (global, national, regional, local) (EPA 2009). It adopts a broad definition of value to incorporate aspects based on human preference (e.g., attitudes and judgments, economic value, community-based value, and constructed preferences) and on the biophysical environment (bio-ecological and energy-based values). In order to capture these dimensions, the report recommends that the Agency (EPA) should not only use economic models and tools, but also methods to capture social attitudes, preferences and intentions, such as civic valuation, a decision science approach, ecosystem benefit indicators, biophysical ranking methods, and cost as a proxy for value (EPA 2009).

A similar recognition of the multidimensional and context dependent nature of valuing biodiversity and ecosystems has been adopted by the UK Natural Environmental Research Council (NERC 2009). Based on the OECD (2002) typology, it distinguishes several types of value (e.g., direct use consumptive value, direct use non-consumptive value, indirect use value, insurance value, option value, and bequest value) and proposes a series of eight pathways for valuation each defined in terms of specific assessment frameworks and disciplinary expertise, but which together aim at presenting a more comprehensive picture of the valuation process. These pathways include valuation of ecosystem services, existence value, recreation and amenities, well-being, direct resource use, genetic and bio-prospecting value, and conservation and energy values (NERC 2009).

In spite of these advances, it is important to recognize some limitations of economic valuation. The inclusion of social and cultural criteria, for instance, while desirable is difficult to attain and constrained by methodological limitations (Shmelev 2008; see also chapter 5), as are studies examining the longitudinal aspects of different attempts to value biodiversity and ecosystem services. In particular, valuation studies face the challenge of recognizing the inter-dependency among the many types of ecosystem services and that of different interest groups competing or associated with different parts of the same ecosystem (Turner et al. 2003).

Considering the issues above, how a middle ground can be found about various implications of economic valuation, including an appraisal of the limitations of what should and should not be valued, are significant challenges before us. Beyond proposing definitive solutions, this chapter aims at contributing to these discussions and to TEEB by raising questions about the trade-offs and challenges of valuation. Valuation represents one particular way of thinking and a perspective that is based on a rational management approach to the environment, and can play an important role in calling attention to the value of biodiversity and to intangible ecosystem services vis-à-vis other forces competing for use and control of particular resources in detriment of others (e.g. standing forest versus land for agricultural expansion). Although this requires some level of objective measurement and some imposition of a value system, it is one way to confront the pressures of market forces which see the environment strictly as a commodity. However, finding a middle ground will imply shifting attention from the question of the value of nature to second-order questions of when, how, and what to value, and whose values count.

The following section continues the discussion of the socio-cultural context of valuation by referring to the trade-offs of valuation as well as its challenges. The discussion of trade-offs of valuation includes (A1) changing notions of property and property relations induced by valuation, (A2) the role of intrinsic values, and (A3) the importance of valuation as a social and economic feedback mechanism. The chapter concludes by focusing on methodological challenges such as (B1) the problem of equity and the transformation of resource value along the value chain, (B2) the growing complexity and inter-dependency of ecosystems and the limitations of valuing ‘islands’ of resources, and, finally (B3) the importance of considering value-articulating institutions.

2 The trade-offs of valuation

2.1 The long-term implications of valuation: the changing concept of ownership and property

In spite of efforts to acknowledge value of biodiversity and ecosystem services as multi-dimensional, contested, and context-specific, exercises of valuation represent an effort to promote some universal notion of the environment, and as such, they carry broad and long-term consequences. By acknowledging the value of benefits derived from biodiversity and ecosystems, perhaps the most important consequence of economic valuation is the way it contributes to change the notion of ownership and property applied to the environment in general, and biodiversity in particular. In this context, to use a reference from Polyani (1944: 73), valuation contributes to create a "commodity fiction." Or, as Sahlins (1996: 411) says, "...a purely Western (construct) that nature is pure materiality." The danger with the commodity fiction is that the commoditized environment becomes a contrived artifact of itself: Ecosystems and biodiversity can be owned and traded in the market system for dollars (Vatn and Bromley 1994: 137).

As such, environmental valuation creates a means for valuing biodiversity monetarily, and thus implies and/or imposes new notions of ownership and property. However, people do not necessarily have previously-defined monetary values for non-market goods (Cummings et al. 1986; Mitchell and Carson 1989) such as biodiversity or biological processes such as carbon uptake, and the process of valuation can trigger negotiations between and within endogenous and exogenous systems of value (Sagoff 1998; Hanemann 1994). This suggests that environmental valuation creates a framework that can induce not only previously ignored monetary appreciation of biodiversity and ecosystem functioning but new utilitarian frames of appreciating them. Gowdy and Erickson (2005) explain that behaviors such as cooperative consensus building and collective decision-making become difficult to pursue in a decision-making framework where only individual preferences count. In other words, the institutional setting influences preferences in a choice situation by activating particular motivations and rationalities (Vatn 2005).

There are two separate though interrelated lines of critique worth pointing here- a. institutional and b. psychological. While cultural and social anthropology points towards flaws in the structural and institutional outlook of economic valuation (Descola's theory points to fundamentally different cultural approaches in conceptualizing human-environment relationship), psychological critique centers on the depsychologizing of human behavior as seen in stated or revealed preference theory where there is little appreciation of intrapsychic or interpersonal origins of human behavior. The relationship between economics and psychology has been quite fraught. Several authors, notably, Sen (1973), Lewin (1996), Johansson-Stenman (2002) and Muramatsu (2009) voice a common criticism that economics has chosen to use psychological theories and constructs in an inconsistent way to explain economic behavior mainly through championing the rational choice paradigm. Sen (1973, 1979) pointed out that despite seeking motivations behind their actions, economists choose to solely focus on the outcome, that is, resultant behavior or have chosen to aggregate the behavioral outcomes as if these were prices and physical quantities. Psychological reasoning, understanding affects, feelings and thoughts underpinning different states of mind, interactions between individual and group, groups and social institutions has been compromised in this process. This is despite the fact that empirical research in behavioral economics, anthropology, psychology and moral philosophy have time and again rejected the standard economic assumptions with respect to people's preferences and behaviors (Sen 1973; Wilk 1993; Kahneman 2003; Nussbaum 2001; Muramatsu 2009) and in terms of theoretical advancement what resulted over time is a corpus of (unsuccessful) nonpsychological preference theory proven to be ineffective, counter-intuitive and limited in explanatory potential (Lewin 1996).

More recent developments in behavioral economics suggest that utility and emotions cannot be divorced from each other, utility arises from emotions and emotions arise from changes and people's judgements and choices have more intuitive than rational- logical origins (Kahneman 2003: 1457; see also Bernoulli 1954). A theory of choice that completely ignores feelings such as the pain of losses and the regret of mistakes is not only descriptively unrealistic, it also leads to incorrect prescriptions that do not maximize the utility of outcomes as they are actually experiences (Kahneman 2000; Kahneman et al. 1997).

A discussion on the implications of attaching new forms and concepts of property to nature and culture (e.g., ethnic affiliation and markers, knowledge systems) is on-going in anthropology (e.g., Dove 2006; Hames 2007; Commaroff and Commaroff 2009), but this is still an overdue discussion in environmental sciences and conservation policy. Since the late 1980s, and concomitant with the rise of protected areas in previously populated regions, numerous policies have focused on granting resource use rights and ownership to indigenous groups and rural populations considered "traditional." These policies are changing significantly local relationships within and between indigenous and rural populations in terms of rights to control, exclude, and derive monetary value based on distinct ancestry, ethnic affiliation, and knowledge of resource use. Martinez-Alier (personal communication, 2007) directs attention to a phenomenon he calls 'fetishism of fictitious commodities' referring to those environmental commodities that are not even in the market and yet are valued in monetary terms.

Hale (2002) has described these processes in Latin America as a form of 'neoliberal multiculturalism', one which has come about during the rise of neoliberalism since the 1980's in part as a response to demands for rights by the culturally oppressed and excluded, in part a move away from class-based politics and more universalist social policies. Despite having opened new political spaces, an over-emphasis on ethnic-based policies are contributing to fragment society into multiple identity groups with few perceived common interests and characteristics. These changes have represented a form of commodification of intangible goods such as ethnic identity usually associated with a repackaged version of the "noble savage" where local populations are expected to behave as stewards of nature (Dove ibid; Hames ibid; Pedrosa and Brondizio 2008). Not only the needs and rights of the local populations ought to be recognized but also their unique interdependence and attachment to the nature needs to be better understood. The association of multicultural policies and environmental conservation has set the stage for competing ownership to natural resources and knowledge systems (Escobar 1998; Kohler 2008). In parallel, critical theory and its influence in disciplines such as anthropology, psychoanalysis, sociology, and ecology have challenged dominance of largely patriarchal, educated, logico-positivist structures of thought and reasoning over voices of weaker, marginalized sections of society (see, Roughgarden 2004; Agarwal 1994; Martinez-Alier and Thrupp 1992; Martinez-Alier 2008) and towards alternative discursive, hermeneutic paradigms (Howarth and Farber 2000; Zografos and Howarth 2008).

Nazarea (1998; 2006) called attention to the value of biodiversity, particularly but not only agrodiversity (e.g., identification of varieties and their specific qualities), as depending in large part on having cultural memory and knowledge associated with it: “Local knowledge and cultural memory are crucial for the conservation of biodiversity because both serve as repositories of alternative choices that keep cultural and biological diversity flourishing.” (Nazarea 2006: 318). The value of medicinal plants, crop varieties or forest resources, for instance within the perspective of bioprospecting, gains meaning as valuable only when associated with knowledge to identify and recognize how to use and manage a resource (Jarvis et al 2007). In other words, these knowledge systems associated with biodiversity are held collectively and inter-generationally and change processually as local systems and practices co-evolve with changing environments (e.g., Pinedo-Vasquez et al 2002).

Bioprospecting programs based on the association of corporations (e.g., pharmaceutical, agronomic), governments and local populations have flourished around the world with the ‘promise of selling biodiversity to protect it’ (Reid et al 1993; Hayden 2003). While the economic benefits of these experiments have been minimal or null in the majority of cases, internal conflicts within and between communities, governments, and corporations have abounded (Greene 1997; Hayden 2003). Some have raised the questions of bioprospecting as another form of colonialism, a “bioimperialism” which appropriates resources and knowledge from marginal groups to powerful corporations using social and environmental discourse (Moran et al. 2001). The key issue, however, is that in doing so, biodiversity and knowledge about it becomes “No longer considered ‘common heritage’, the pre-CBD paradigm that provided open access to bioresources.” (Moran et al. 2001: 501). It has become clear that the commodification of knowledge which evolved historically from a collective base do not lend themselves to the application of `conventional’ legal-economic tools of property rights, such as industrial patent, intellectual property, and royalties. Examples in Mexico (Hayden 2003), Peru (Greene 2004), and South Africa (Commaroff and Commaroff 2009), among many others, illustrate some of the trade-offs of the valuing and selling to protect approach, at least when it comes to the distribution of benefits. The unfolding lessons of bioprospecting programs started during the 1980s and 1990s can serve as powerful examples for other programs of economic valuation to reflect on their long-term and potentially negative implications.

2.2 Intrinsic value of nature and value articulating institutions

Among the values of ecosystems are cultural values perceived by specific cultures (e.g. the belief in holy trees). But, all values are culturally constructed and contextualized. Values are institutions and as such (contribute to) define behavior. Values can be made visible by applying specific valuation methods and the valuation methods themselves are socio-cultural constructs which define the rules for eliciting or articulating values. Choosing the socio-cultural context of valuation also implies a choice of the respective valuation method. Valuation methods themselves are designed in a way and emerge from the understanding of what values are, or should be and how they can be elicited. Valuation methods, for example imply certain models of humans, nature and their interactions and they define whether values are revealed, discovered, constructed or evolve during the process of valuation (Vatn and Bromley 1994). Vatn (2005) refers to valuation methods as value articulating institutions. Values for the same ecosystem service therefore vary across institutional settings.

The question of the value of nature also raises the opposite question of the nature of values (Gatzweiler 2003). Values, as well as norms, beliefs and conventions of society are an essential part of our culture. Values derive from worldviews and fundamental perceptions of a society of, e.g. what is right or wrong, good or bad, valuable or worthless. They are deep manifestations of a culture of which not all can be directly observed or predicted through models of rational choice (Wilk and Cligget 2007). A large number of empirical studies based on experimental games as well as experimental social psychology have shown that apart from being egoistic utility maximizers, as assumed to be the case for the *Homo economicus*, "people tend to be more altruistic than the economic model predicts" (Gowdy et al. 2003: 469) and that they act both selfishly and cooperatively (Alesina and Ferrara 2000; Fehr and Tougareva 1995; Gintis 2000; Güth and Tietz 1990; Manski 2000; Nowak et al. 2000; Ostrom 1990; Etzioni 1986; Caporael 1997; Kahneman 2000; 2003).The model of *Homo reciprocans* presented by Bowles and Gintis (2004) suggests that people behave altruistically to those who reciprocate their altruistic behavior. Depending on whether they perceive the behavior of others as being beneficial or harmful, they will respond in kind.

Gowdy et al. (2003), however, have presented yet another type of behavior observed in a rural Nigerian village, where fairness was an important predictor of economic behavior, but not retaliation. Their case demonstrates that non-cooperative behavior elicits a cooperative response and that "retaliation is much less common in traditional cultures than in Western societies." An important implication of those findings is that behavioral differences among cultures are large and they are often correlated with group norms and values, and not with attributes of the individual. This should be clearly articulated in the context of our attempt to value ecosystem service and biodiversity according to a model evolved from the very particular cultural tradition of the industrialized world.

The view of nature and humans being distinct from each other, as discussed above, shows itself in the neglect of intrinsic values in economics – values of nature simply for the sake of its existence, independent of any current or future usefulness to humans (Gatzweiler 2008). The Newtonian conception of reality, however, has fundamentally changed with quantum physics, the philosophical substance of which tells us that no clear distinction can be drawn between observer (subject) and observed (object), or, human and nature. This has consequences for human cognition, because it entails that there is a close connection between human software and hardware: the way people perceive their environment (software) and the way they measure, value and construct it (hardware). From that holistic perspective, "what we observe is not nature itself, but nature exposed to our method of questioning. […] The observer decides how he is going to set up the measurement and this arrangement will determine, to some extent, the properties of the observed object." (Capra 1991: 140; see also Heisenberg 1958).

In this context, the methods used to elicit values define the values actually elicited. If individuals are asked about their willingness to pay for ecosystems and biodiversity, it is likely that people actually state their willingness to pay for ecosystems and biodiversity and the method requires the individual to articulate its values according to a consistent logic and specific rationality. Other methods allow for communication, deliberation and value statements then emerge from a social process.

The issue of intrinsic values is helpful to reflect on the relationship between nature and humans. It proposes that nature has value in itself and is valued as an end in itself, independent of its usefulness to achieve some higher end. The question whether intrinsic value can or should exist or not, directly relates to how we perceive human-nature relationships and the way people relate to nature is not only reflected by their actions but also by the rules they apply to articulate values for nature. Therefore, acknowledging intrinsic values of nature acknowledges the fact that people are part of nature and "it is how we choose to perceive people and biodiversity that determines choices of how to (value and eventually) conserve biodiversity" (Gatzweiler 2008).

The point made here is that the approach to eliciting values from people for ecosystems reflects understandings, perceptions, and normative stances of what values are and how values are generated and held: the pre-analytic conceptions of those asking questions. And just these pre-analytic conceptions define the values the researcher wishes to discover or create. Two extremes are to ask individuals about their willingness to pay or allowing people to deliberate. Asking individuals about their willingness to pay thereby reflects different pre-analytic conceptions than allowing them to deliberate, whereas former assumes that people:

- hold these values in advance or can easily generate them
- have sufficient information and understanding of what they are valuing
- can decide (alone) on the values they attribute to ecosystems
- behave according to the cost-benefit rule
- value consistently
- value according to individual rationality

On the other hand, deliberative valuation methods do not assume pre-existing values for ecosystems and biodiversity. Given the fact that values are part of the institutional and cultural context people live in and that this societal context has co-evolved over long time periods it is likely that values are not held in advance and that people need to communicate and deliberate on issues which require valuation. In such deliberative processes values emerge from a communicative social process.(O'Connor 2000; Zografos and Paavola 2008). Commonly known techniques such as Participatory Rural Appraisals (Chambers 1991) Citizen Juries or Roundtables can be suitably modified to facilitate these processes.

The foregoing discussion makes a case for environmental valuation as value articulating institution (Jacobs 1997), i.e. a framework that is invoked in the process of expressing values and which influences which values come forward and what sort of conclusions can be reached on the basis of those values. Vatn (2005) defines a value articulating institution as a "constructed set of rules or typifications" which specifies the conditions under which values will be expressed such as what type of data will be deemed relevant (e.g. environmental valuation considers only monetary bids as relevant data), who participates in valuation (similar concerns raised by feminist economics, Agarwal et al. 2005) and in what capacity (e.g. environmental valuation asks individuals to participate as consumers). He further explains that different value articulating institutions "tend to give different outcomes or preferred solutions", which implies that "the choice of such institutions is certainly non-trivial" (ibid: 211).

As a value articulating institution, environmental valuation is not particularly inclusive of plural environmental values, given that values of some ecosystem services cannot be monetarized. However, as also discussed in chapter 5, alternative value-articulating institutions such as multi-criteria evaluation and deliberative processes (e.g. citizen juries, etc.) try to reflect environmental values and motivational plurality (Table 1). These alternatives also attempt to consider the criticism towards environmental valuation that people may want to participate as citizens instead of consumers in environmental decision-making (Sagoff 1988).

Table 1 Value articulating institutions and respective normative and epistemological stances (Source: Modified from O'Connor et al 1998)

Value articulating institution	Normative and epistemological stance
Contingent valuation method	Cartesianism: Value is pre-existing and needs to be discovered. Separation between values and facts, human and nature. Substitutability between money and ecosystem goods and services. Values are revealed.
Deliberative or social process methods	Democracy stance: value is constructed in social processes. Previously unknown values evolve from deliberation and debate. Prioritizes each member of society to contribute to knowledge and judgment.
Multi-criteria methods	Complexity: Value understood in terms of ranked importance. Irreducible plurality of analytical perspectives for a stationary enquiry.

Deliberative methods stem from an awareness of the need to acknowledge and legitimize plural values in public policy and decision-making. Deliberative democracy scholars require that beyond other outcomes policy generates a public domain where reflection upon preferences is stimulated in a non-coercive manner, by means of information provision and deliberation (Dryzek 2000). As deliberative democracy's aim to pursue such public spheres is in tune with environmental value plurality, deliberative forums (e.g. citizen juries) seem to provide a desirable model of a value articulating institution. However, the potential of deliberative decision-making has skeptics. For example, advocates of deliberative planning are reproached for paying "insufficient attention to the practical context of power relations in which planning practice is situated" (McGuirk 2001: 196). Likewise, others argue that "a deliberative and democratic praxis of sustainability may be effective only if and when underpinned by substantive changes to the exercise of power and leadership" (Stratford and Jaskolski 2004: 311). Similar concerns have been raised as regards biodiversity management through deliberative decision-making processes (O'Riordan 2002).

Valuation scholars have attempted to integrate deliberative processes in environmental valuation by means of developing deliberative methods of environmental valuation (Sagoff 1998). This practice can be seen as a response to criticisms towards contingent valuation (CV); those criticisms postulate that environmental value is a group value and should not be sought as an aggregate of individual values. The practice also tries to take on board criticisms that environmental preferences do not exist ex-ante but are socially constructed (Vatn 2005) and that values are sensitive to changes in issue framing and information brought to the attention of the public during the process of value elicitation (Slovic et al. 1990). Basically, deliberative valuation tries to turn the value elicitation process into a preference-constructing process in order to deal with the issue that people do not hold pre-determined preferences towards the environment and that such preferences should be well-informed and deliberatively derived (Zografos and Howarth 2008). However, critics of deliberative environmental valuation point out that in practice it has been applied as a means for justifying stated preference methods by adding often superficial forms of deliberation or discussion, and that in essence the relevant studies establish that the economic model they use is unsuitable for understanding particular sets of social values as regards the environment (Spash 2008).

2.3 Valuation as a feedback mechanism

Exercises of valuation can play an important role in calling attention to the value of biodiversity and to intangible ecosystem services vis-à-vis other forces competing for use of particular resources to the detriment of others. Although this requires some level of objective measurement and some imposition of a value system, it is also a way to confront the pressures of market forces which approaches the environment as commodity. Further, valuation of ecosystem services can create incentives for land use change, such as promoting what has been called an agroecological transition aiming at reconciling the value of production and environmental services (Mattos et al. 2008). While several parts of this chapter have called attention to the potential negative implications of economic valuation, its value as a decision making and awareness mechanism to society are also clear. One can argue that in the long run, this approach actually will lead to the internalization of the environment into western thinking and economics. In this context, valuation methods can serve as a mechanism to provide feedback in a system where production and consumption, trade and exchange are so distant and complex that they undermine perceptions of the impacts of habits and behavior on the environment (Moran 2006; Wilk 2002). One can see these processes as a form of regulatory adaptation where behavioral responses within particular cultural and social contexts are taken progressively to cope with environmental changes perceived as detrimental (Moran 2000). In this context, valuation can be seen as a feedback mechanism which confronts the problems of market demand for commodities and lack of accounting for externalities with the same tools and language, i.e., values and costs. However, the processes that mediate perceptions of value and actions to conserve biodiversity and ecosystems carry a time lag between behavioral responses at the levels of the individual and whole populations, in which the actions of the former can be overwhelmed by the inactions of the latter. In this context, as other processes affecting society, the impact of valuation on behavioral changes are functions of cultural context (e.g. perceived notions of value, whether changes are culturally accepted), society and economics (e.g. degree of participation of the larger society, available institutional arrangements facilitating collective action), and the perceived environment benefits (e.g. availability of resources or access to desirable landscapes) (Brondizio and Moran 2008). Furthermore, adaptation to environmental change depends on forms of institutional arrangements that facilitate these activities within and across levels.

The socio-cultural construction of economic value is not static, but evolves in a processual way as behavioral actions respond (or not) to feedbacks. It coevolves with changing perceptions of society's environmental reality (Norgaard 1987; 1984: 165). This process of co-evolution is underpinned by a cognitive performance of mutual specification and co-determinism (Varela 1999, Maturana and Varela 1928): humans bring forth their own domain of (environmental) problems and solve them according to their ability to order interactions with nature. This process of ordering interactions between humans and nature is also facilitated by institutions, because "institutions pattern lives" (Tool 1986: 51). Therefore, ecosystems are degraded and biodiversity is lost, attitudes and values towards nature (must) change. The institutions according to which people pattern their lives will then change as a consequence, but the timelag may be long.

The values attached to ecosystems and biodiversity (or anything else), are not only determined by a constructed ethical environment and the respective institutions. They also depend on social emotions and feelings. The extinction of the blue whale might be deemed economically rational by some (Clark 1973). Ethically and culturally, however, the extinction of blue whale would make many people feel incensed and react in extremely angry ways. Because of the complexity of the issue, those people may not be able to reason scientifically and logically why the blue whale should not be hunted until extinction; they could merely express their unease about it. Both stances employ their very own ethics: The economic stance is based on an ethics of individual rationality (which also defines "good" and "bad" decisions) and the "ethical stance" is one that is based on some feeling of what is "good" and "bad".

Damasio (2003: 162) identifies feeling as the "embryo of ethical behavior" and part of "an overall program of bioregulation." He defines feelings as homeostatic devices to keep the body-brain system in balance, just like institutions are rules to keep social and socio-ecological interactions in balance. Damasio says that ethical behavior depends on the working of certain brain systems which are not exclusively dedicated to ethics but also to biological regulation, memory, decision-making, and creativity. On those grounds, the role of feelings can be tied to natural, life-monitoring functions: "Ever since feelings began, their natural role would have been to keep the condition of life in mind and to make the condition of life count in the organization of behavior. Our life must be regulated not only by our own desires and feelings but also by our concern for the desires and feelings of others expressed as social conventions and rules of ethical behavior … feelings remain essential to maintaining those goals, the cultural group considers unavoidable and worthy of perfecting. Feelings also are a necessary guide to the invention and negotiation of ways and means that somehow, will not clash with basic life regulation and distort the intention behind the goal. [They] remain as important today as when humans first discovered that killing other humans was a questionable action." Feelings are important for decision making to enable people to deal with the uncertainty inherent to all complex decision making situations. Although abstract, they are located where communicative interactions, social rationality and complex system properties are taken into account, thus, as an important aspect of human behavior, they remain a central component for valuation.

3 The challenges of valuation: ecosystems, biodiversity and level of analysis

The scope of this section is to present, on the one hand, the main challenges of addressing various levels of analysis using valuation methodology and accounting, and, on the other hand, the required attention to complexity embedded in resource use systems today. It is divided into 3 parts, each of which is built upon issues raised above.

3.1 The problem of transformative value and economic return of resource use

Indigenous and rural populations, although often considered stewards of biodiversity, share an unequal position, usually at the lower end, of larger commodity chains of resources. Around the world, the value of resources increases along the market chain usually far from their areas of origin, thus creating unequal distribution of benefits and weak incentives for conservation and management. As a resource moves from a state of raw material to various levels of industrial transformation, its economic values are increasingly attached to market symbols aimed at different groups of consumers (Brondizio 2008). This is a classic situation for many valuable resources coming from tropical forests or aquatic systems around the world. On the one side, the producer of tropical forest fruits who manages standing forests receives a few dollars for a basket of fruit while on the other side a consumer in the United States pays high prices for products which in some cases contain only traces of the same fruit, but which holds splashing advertisement and claims about sustainable development. Thus, what parameters should be used to value specific resources? Which basis can we use to estimate the value of a resource as it changes in price as much as 70 folds along a commodity chain? If valuing biological resources is a tool to improve in situ conservation, it assumes that local stakeholders have sufficient incentives to maintain a given ecosystem against other competing uses. The symbolic value embedded in resources as commodities mediates its economic value along a commodity chain (e.g., Appadurai 1996; Haugerud et al. 2000; Brondizio 2008). The economic value of forest resources, for instance, becomes dependent not only on their demand as raw material, but the level of industrial transformation and, most importantly, the symbolic meaning attached to their marketing as end products to consumers. The pragmatic dimension of this discussion, as illustrated by the case of acai palm fruit of the Amazon presented in Box 1 and Ethiopian wild coffee in Box 2, is the importance of aggregating value to resources locally as a form of creating incentives for local management vis-à-vis conversion to other uses because of market pressures. The Amazon illustrates well this tension. The combination of limited global availability of arable land associated with increased demand for vegetable (e.g., soy bean) and animal protein (e.g., beef), vast land availability, government priority to export surplus, and the low value of forest resources at a regional level explains the majority of Amazonian deforestation during the past two decades.

Box 1 The boom Açaí palm fruit in the Amazon

There is possibly no better example of an economic prospect for reconciling forest conservation and development Amazônia than the case of the açaí fruit (*Euterpe oleracea* Mart.) production system (Brondizio 2008). Emerging from the initiative of local producers to supply a growing market demand for açaí fruit, using locally developed technology and knowledge with respect to forest management, açaí fruit production embodies the social and environmental principles that permeate the discourse of sustainable development for the Amazon region. At the same time, the formation of this production system poses important questions concerning the spread and duration of benefits resulting from booming tropical forest economies. To what extent are production and market opportunities to value forest resources diminished by a history of socio-cultural prejudice, land tenure insecurity, and differential access to economic incentives? The expansion of the açaí fruit economy occurs as a combination of both endogenous and exogenous factors associated with the region as a whole, and in association with its consumption basis. These include rural out-migration and urban expansion since the 1970's, the organization and marketing strategies developed for the export of other Amazonian fruits during the 1980's, and the growth of the "green products" industry during the 1990's. The growth of açaí fruit consumption is driven by various claims relating to its healthy and invigorating qualities, rainforest conservation, respect for Indigenous causes and products, and its representation as an icon of the sustainable development agenda proposing alternative forms of land use in the Amazon. Açaí fruit's secure position at the regional level as a staple food favorite, as well as its expanding national and international markets, has transformed açaí fruit into a symbol of cultural identity and regional pride for Amazonian small farmers, particularly in the Amazon estuary. Today, it has been industrialized into a range of products of popular consumption, such as yogurts, concentrated juices, ice creams, energetic beverages, vitamin pills, as well as products such as shampoos and soaps.

Overall, for most of the history of açaí economy producers have received better prices than the average price of most agricultural and husbandry products of the region. Analyzing the evolution of prices, it becomes clear that açaí producers had an incentive to manage forests for açaí production. As a result, during the past 30 years, the Amazon estuary as experienced, contrary to elsewhere in the Amazon, a forest transition, i.e., high rates of regrowth, increasing forest cover, and minimum deforestation. Emerging from a local rural economy, the açaí fruit industry is now functioning as a complex multilevel economic structure. As part of this process, forest managers and producers of açaí fruit negotiate a position amid regional and international investors and companies, although suffer from the lack of infrastructure to commercialize their product and incentives to participate on value aggregation. This creates a paradoxal situation whereas the açaí fruit economy continues to grow in scale, but the proportion of revenues retained locally is increasingly smaller. Although producers have been benefiting from the expansion of this market, they have been unable to participate on new sectors of the economy associated with the commercialization and control of fruit stock, its transformation, and its value aggregation along the chain. Producers suffer from the stigma of extractivism and the invisibility of their intensified management and production in standing forests (still widely referred as an extractivist system), a situation which continues to maintain them as suppliers of raw material (Brondizio and Siqueira 1997). New entrepreneurs and large regional producers have come to occupy the most profitable niches of the market and assume greater control over production, commercialization, processing, and marketing. Estimates of the current economic impact of the açaí fruit market in the region and abroad range from R$100 to 500 million/year, possibly much larger depending on how, what, where, and how far the commodity chain one counts (Brondizio 2008).

Most of this economy, however, aggregates value away from production areas. For instance, the value of acai fruit pulp resulting from the harvest of one hectare of managed forest at the farmer's gate (i.e., fruit in nature) ranges from around US$ 1,000 to US$1,200. The same amount (in equivalent processed pulp) will increase 20 to 50 folds (depending on the end product) when reaching consumers in southern Brazil and up to 70 folds or more (depending on the end product) when reading international consumers (Brondizio ibid.). Further, the increasing competition from new areas of production and corporate plantations seeking to control supply are leading, progressively, to increasingly to monocultural systems (vis-à-vis forest management). The lack of transformation industries installed locally and accessible to producers that could help to aggregate value locally (to producers and municipalities) is progressively decreasing incentive for managing and maintaining diverse standing forests where small farmers manage several species vis-à-vis other land uses.

Box 2 Commodity and symbolic values for wild coffee from Ethiopian forests

Despite Ethiopia being the largest Coffee Arabica producing and exporting country in Africa, 98 percent of the national coffee production comes from smallholdings which are less than a hectare in size and 95 percent of that coffee is produced in forest-, semi-forest and garden systems. The commodity chain involves producers, cooperatives, exporters, importers, roasters, retailers and consumers. Coffee is bought and sold as a tradable commodity and the farm gate prices of coffee in Ethiopia is connected to price fluctuations at the New York Commodity Exchange. Although some specialty coffees, like wild forest coffee, are connected to retail prices, the largest price margins are still achieved between the roaster and the retailer and between the retailer to the consumer. Once the coffee has reached the consumer it is no longer just a commodity which is valued for its quality as raw material, and by the forces of supply and demand. It is now a lifestyle product for which consumers are willing to pay because it responds to their needs, wants, beliefs or convictions. Whereas in 2006/7 an Ethiopian farmer in Yayu received 0.5-1 USD (=0.4-0.8 Euro) per kg green wild forest coffee, when he delivered it to the cooperative, 1 kg of packaged, roasted wild forest coffee is now sold for 38 Euro/kg. If a kg of coffee is sold in form of 100 warm cups of coffee á 2-3 Euros/cup, its value has already increased to 200-300 Euro/kg.

On the other hand, although valuing biodiversity can be a tool for in-situ conservation, the increasing economic value of forest resources along the value chain can also be a disincentive for biodiversity conservation. Seyoum (2009) shows that higher incomes for households in Ethiopia close to coffee forest areas, can be an incentive to intensify coffee management inside the forest and thereby reduce the wild coffee and forest diversity. However, the discussion of whether intensification of land use leads to deforestation is still poorly understood (Angelsen and Kaimowitz 2001), in part because this relationship is mediated, on the one hand, by the role of markets and their distributive benefits (i.e., the degree of value aggregation at a local level), and on the other hand, by the effectiveness of institutions regulating use of resources. The two examples presented here illustrate well the problem of value aggregation and the counter-forces of intensification.

The recognition that value change along the commodity chain has implications for the distribution of benefits and affects the level of incentives for conservation represent an important methodological challenge for economic valuation. The inability of conservation and development programs to create incentive systems to aggregate value locally (and thus employment in rural areas) represents a widespread problem not only for developed countries, but around the world. Initiatives such as certification and terrier recognition are growing with diverse success in different parts of the world. In general, however, the lack of policy frameworks to promote local value aggregation and reduce distances between producers and consumers fuel an economic logic whereas the market for monocultural plantation and/or cleared land is many-fold higher for the [valuable] resources of standing forests or the rich agrodiversity passed down through generations.

3.2 Complexity and functional inter-dependencies underlying valuations

Economic valuation is a complex, spatial and institutional cross-scale problem (Turner et al. 2003). As also pointed out in chapter 5, section 3, values of ecosystem goods and services differ with changing ecological features and with differing size and characteristics of groups of beneficiaries. Whereas recreational values of a site may be valued for its direct use at local scale by visitors of the site, the value of high levels of biodiversity may be valued for its option, bequest, existence and altruist benefits at a global scale by the global community.

Many efforts focusing on particular parts of ecosystems or species, such as the creation of protected areas, while effective at one level, lack the scope to control the pressure of commodity markets for land resources surrounding them. As such, and depending on their biophysical context, they are limited to capturing the linkages and vertical interplay created by a growing functional interdependency of resource use systems nested within larger ecosystems (Young 2006; Brondizio et al. 2009). Increasingly common around the world, as described at the introduction, 'islands of protected ecosystems' are nested and affected by systems at higher or lower levels, and thus have substantial long-term limitations to guarantee conservation. Furthermore, they illustrate the importance of understanding the diversity of cultural perspectives to the environment. Take for instance the case of an indigenous group which values and has successfully protected forests within a given territory, an area/ecosystem which is, however, nested within a larger watershed. Assume the larger watershed is occupied by very different groups of people who have very different perspectives of the environment and who are closely responding to global markets for agricultural commodities. The result is that rampant deforestation outside reserves will systemically undermine the [protected] environment through water pollution, soil erosion, and forest fire, including the possibility of reaching thresholds which may lead to unpredictable ecosystem changes. As the authors describe, recently arrived farmers may see the forest as a threat and the environment as sets of resources to be transformed. On the other hand, the environment as a whole is an intrinsic part of indigenous cosmology and an organic part of their economy. Indigenous groups carry detailed intergenerational knowledge about forest and water resources, cultural attachment to place, and customary rules of use and resource appropriation which tend to hinder members of the group from carrying out short-term and large-scale transformations that are characteristic of large (and small scale) and corporative farmers aiming at taking immediate opportunities of commodity markets. The authors stress that "we should build social capital that enhances the long-term sustainability of natural capital at multiple levels on scales of relevance to particular ecological resources." In other words, it points to the role of institutions in facilitating cross-level environmental governance as an important form of social capital that is essential for the long-term protection of ecosystems and the well-being of different populations. (Brondizio et al 2009: 258-259). This scenario illustrates the challenges of conservation and development in the Brazilian Amazon during the past two decades and currently. The creation of a record number of protected areas and indigenous reserves, today corresponding close to around 30% of the region, happened concomitantly with the period of highest rates of deforestation ever observed. Increasingly, the region is observing the formation of `islands' of forests and forest fragments (Carneiro Filho and Souza 2009). This scenario represents the trajectory of many regions around the world and raises the issue of social and environmental interdependency and the limitations of valuation methods which may account for valuing resources at one level, but neglect other levels affecting its long term sustainability.

The situation of vertical interplay of institutions (Young 2006) representing groups competing or cooperating for authority over resources requires one to look at questions of subtractability (i.e., whether resource appropriation by one user reduces availability to others) and exclusion (i.e., how costly it is to keep potential beneficiaries out of the benefit stream) from a multiscale perspective (Brondizio et al. 2009). Local forms of use and regulation of a resource (e.g., based on customary rules of use and exclusion), while potentially effective at a local level, are affected and in some cases overwhelmed by resource use in a different part of the larger ecosystem. As called attention by the MA, one of the biggest challenges of contemporary environmental governance is to promote conservation outside protected areas (Bhattacharya et al. 2005).

Along these lines, valuing resources and protecting an ecosystem requires attention to the value of connectivity at a landscape level. An overemphasis on conservation focused on carbon storage, for instance, with metrics based on stocks of carbon may downplay the role of connectivity of habitats, habitat and species diversity, or the water quality within a watershed. Interconnected social-ecological systems are dynamic, thus requiring constant monitoring and institutional adjustments, but most institutions and forms of incentives are designed to be applied at a given level. Görg (2006) calls attention to multi-level decision making as a pressing issue for environmental governance. He proposes the concept of landscape governance as an approach based on the notion of society relationship to nature to bridge what he calls the "politics of scale" ('socially constructed spaces') and the biophysical interconnection between places. In this context, the challenge for valuation is to function as part of a larger process of co-evolution and adaptive management which stresses, on the one hand, the value of flows and connectivity within and between ecosystems, and on the other hand, facilitates the dissemination of knowledge and responses of institutions and social groups across levels.

3.3 How to choose how to value

One relevant question for decision makers is about how to decide which valuation method is to be used to guide decisions. According to Arrow (1963) and Sen (1970b), the social choice problem is to make decisions for a society composed of a variety of members having non-identical interests and values.

Ostrom (1990), McGinnis (1999), Ostrom et al. (1994), among others, have tried to answer the question of how to design institutions for the governance of complex resource regimes (such as water, forests, or knowledge). Their proposed design principles for successful governance of common pool resources have proven very useful to policy making, particularly when applied to specific local level situations. They were guided by a rule defined by Ashby (1952) formulation of the 'Law of Requisite Variety'. This law says that any regulatory system needs as much variety in the actions it can take as exists in the system it is regulating. Ostrom and Parks (in McGinnis 1999: 284) concluded: "the more social scientists preach the need for simple solutions to complex problems, the more harm we can potentially cause in the world" (see also Ostrom 2009).

There is today considerable evidence that putting a monetary value of an environmental change is a cognitively very demanding task for which people tend to use various simplified context-depending choice rules, thereby implying that the responses are often difficult to interpret (Schkade and Payne 1994; Vatn and Bromley 1994) and psychologists like Kahneman and Tverskry spent nearly 40 years trying to show that people have developed preferences for very few familiar good and for most circumstances employ various heuristic choice rules (Johansson-Stenman 2002).

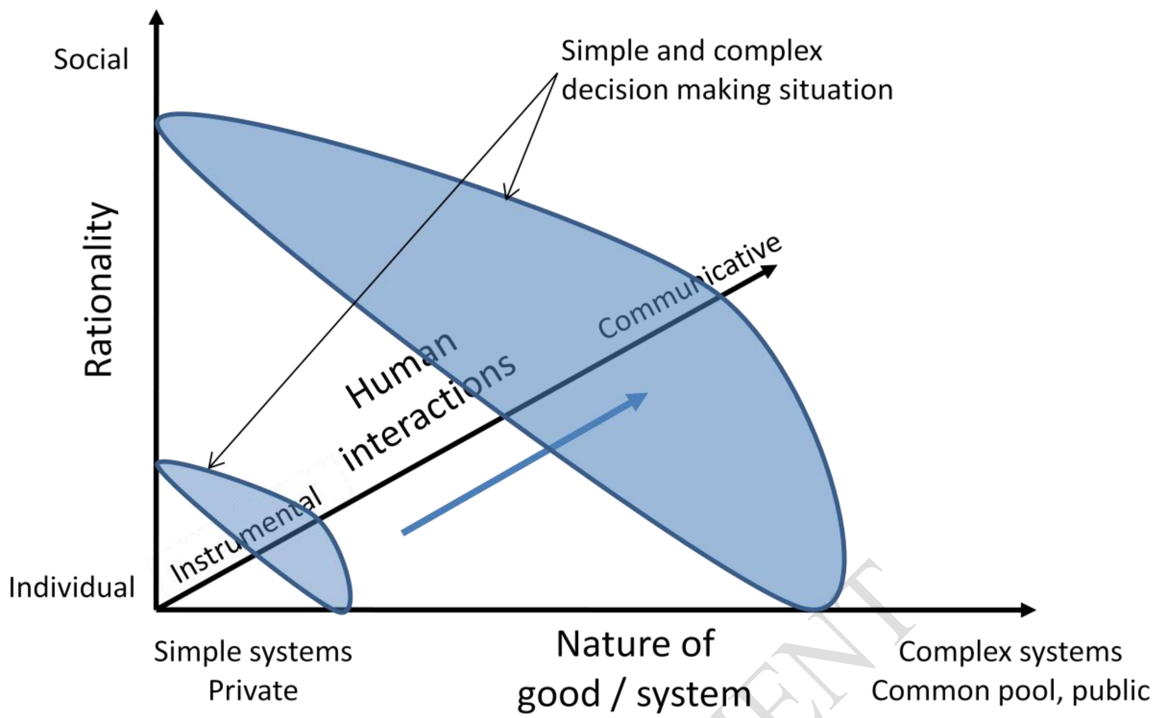

Figure 1: **Dimensions for choosing the valuation method**
Source: adapted from Vatn (2005: 419).

Making decisions in situations of high complexity and incomplete knowledge is characteristic for decision situations related to ecosystems and biodiversity. According to Ashby's law such situations require methods which are able to capture value plurality, ecosystem complexity and biodiversity. This would require a move from aggregating individual values to reasoning over a common set of priorities. Because "…to handle the common goods aspect social rationality and some form of communicative process must be taken in account. It is the only institutional structure that can be true to the choice problem at hand." (Vatn 2005: 421). Strategies for the management of complex systems have been developed (Malik 2008) (Figure 1).

Applied to the complex problem of biodiversity conservation this would require matching the complexity of the problem situation with the organization of public engagement, which is (principally) able to capture as much variety (value plurality, types of rationality, etc) as in the system it aims to conserve. Neglecting socio-ecological complexity (e.g. by limiting discussion to the rationality of economic man and the market system) leads to increasing system vulnerability and increasing danger of system collapse. Applying valuation methods which apply simplified models of the complex systems being addressed, by, e.g. monetary valuation, would consequently not only be less useful, it would reduce value plurality from the start.

O`Connor and Frame (2008) therefore suggest, "that the logic of valuation [...] is: 1. make the proposition to sustain/conserve the forms of community or environmental features in question (e.g., avoid the production of toxic wastes, preserve a designated forest system or other feature of nature), and then, 2. investigate what commitments this does or might entail for—and on the interfaces between—the various communities of interest involved." To engage methodologically with this hydra-like problem, O'Connor and Frame, (ibid) introduce a sequence of strong dialectical simplifications. First, they propose two main types of thresholds beyond which assessing trade-offs or the consequences of choices on the basis of monetary measures alone are of questionable pertinence. Either the estimation is scientifically very difficult, or the proposition of a "trade-off" implied by the opportunity cost considerations is deemed morally inappropriate (Figure 2).

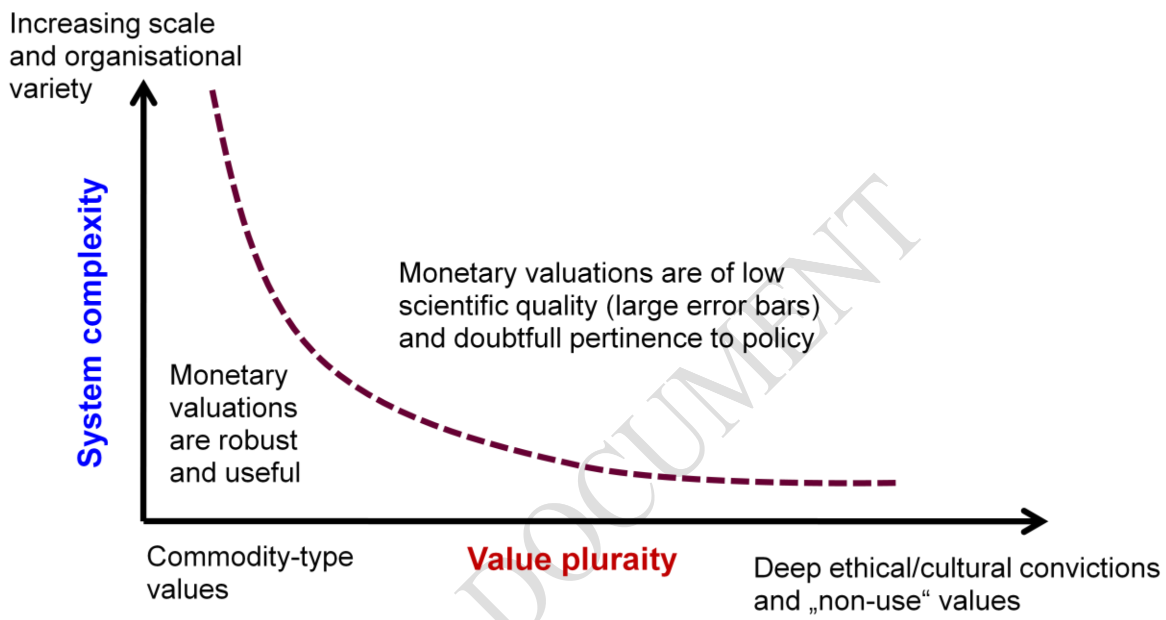

Figure 2: **With increasing system complexity and value plurality, monetary valuations become of low scientific quality and doubtful policy relevance**
Source: Redrawn from O'Connor and Frame (2008).

Recognizing the fact that valuation methods are 'value articulating institutions' and that the choice of a method can strongly influence the outcome of a valuation exercise and thereby actual behavior, supports Atlee's (2003) argument that people are co-producers or 'co-creators' of institutional change: (global) environmental change is a collective process, the consequences of which individuals can not comprehend in its entire range, depth, and detail. Therefore, better ways are needed to perceive and reflect the state of the earth and to facilitate integration of the individual diversity of perceptions and values. Atlee defines co-intelligence as a human capacity and ability to generate creative responses.

And therefore, the choice of value-articulating institution (=valuation method) will define the outcome of the valuation exercise. For instance, :

- the private good side (instead of common pool and public features) of ecosystems and biodiversity,
- simple (instead of complex) systems,
- the individual and egoistic (instead of social) side of human behavior and rationality and
- the instrumental (instead of communicative) type of human interaction,

will bring forth values a world as understood and seen by the eye of the beholder. That means, the way ecosystems and biodiversity are perceived determines the way they are valued, and the way they are valued determines the human interaction with and (mis)use of the natural environment. The decision situations with ecosystems and biodiversity are not simply a matter of value exclusion, meaning that some values of ecosystems and biodiversity have not been considered. The decision making situation is about applying (what one thinks is) the right valuation frame of reference for the valuation of ecosystems and biodiversity. Valuing biodiversity and ecosystems in order to conserve it require more than attaching additional values to nature by appreciating its goods and services. The choice of the value-articulating institution becomes more important than valuing nature in order to prevent market failure. That is, one may assume that if markets do not fail, ecosystems and biodiversity will be conserved. We do not know that and we cannot know that for sure. Therefore one may argue that it is more important how we value than which value we attach to nature.

4 Final Remarks

Trying to put a value on biodiversity and ecosystem services involves tradeoffs. The broader literature on economic valuation recognizes these challenges and problems, and the multi-dimensional and contested nature of these approaches. The social sciences literature calls attention to some pitfalls and the potential long-term implications of economic valuation. Many challenges remain ahead, among others: the difficulty to account for inter-linkages between different ecosystem services, the lack of tools for cross-level valuation and mechanisms to promote value articulating institutions, and the limitation of valuation tools to promote equity distribution and value aggregation to resources and ecosystems at the local level. On the other hand, properly used, economic valuation has the potential to serve as a tool of awareness and as a feedback mechanism for a society which has distanced itself from the resources it uses and from the impacts of its uses on distant ecosystems and people.

Economists know and have known all along (not just since the rise of ecological economics) that their value-articulating institutions are not all-inclusive. Therefore, valuation is essentially a matter of choosing how to perceive the human being itself, how to perceive human’s place in nature, and how to perceive nature itself. This is because the way we perceive our natural environment determines the way we value and change it. One way of incorporating a multilayered understanding of human-environment relations and understanding the value and motivational linkages between the two is to address the large gap that exists between the language in which the preference of the people for ecosystem services is elicited and the language in which people feel more at home. The languages of research and policy show similar dissonance. The more the discourse moves away from the common lives and real life concerns to abstruse quantification and reductionism, the more people are likely to give preferences that are fudged and confused as much as these are confusing, merely because the choices we offer are far from adequate (Kumar and Kumar 2008: 814). Valuation approaches aiming at addressing complex socio-ecological systems require attention to the challenge of understanding problems of credibility, saliency, and legitimacy at the intersection of different knowledge systems and access to information at different levels and by different groups (Cash et al. 2006). In this sense, valuation mechanisms should be seen as part of a broader range of diagnostic and assessment tools and political-institutional mechanisms that facilitate the understanding of complex socio-ecological systems (Ostrom 2009), as well as coproduction, mediation, translation, and negotiation of information and knowledge within and across levels (Cash et al. 2006; Brondizio et al. 2009). The main lesson that comes across when one reviews valuation literature is to avoid an 'one size fits all' approach, or as Ostrom (2007) puts it when proposing a framework for the analysis of complex social-ecological systems, we need to move beyond panaceas.

Economic valuation may contribute to address our inability, reluctance or ideological intolerance to adjust institutions (also those which are value articulating) to our knowledge of ecosystems, biodiversity and the human being. As such, it can contribute to more inclusive economic accounting and planning, and a more inclusive view of non-human beings. In other ways, however, it can also contribute to separate people and nature further apart by simplifying its meaning and value to human societies. In this balancing act, one hopes valuation approaches will not be taken as panaceas, but as tools which may contribute in the long-run to internalize a respect for nature into western cosmology and social life.

[i] For all purposes, we use environment and nature as interchangeable terms, unless referring to their use by particular authors. For a comprehensive review of these terms, particularly the concept of nature, see the volume edited by Ellen and Fukui (1996), in particular, Ellen (1996).

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Chapter 5

The economics of valuing ecosystem services and biodiversity

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Contents

Key messages

- In the Total Economic Value (TEV) framework, ecosystems may generate output values (the values generated in the current state of the ecosystem, e.g., food production, climate regulation and recreational value) as well as insurance values. The latter, closely related to "option value", is the value of ensuring that there is no regime shift in the ecosystem with irreversible negative consequences for human wellbeing. Even if an ecosystem or some component of it currently generates no output value, its option value may still be significant.

- Estimating the value of the various services and benefits that ecosystems and biodiversity generate may be done with a variety of valuation approaches. All of these have their advantages and disadvantages. Hybridizing approaches may overcome disadvantages of particular valuation methods.

- Valuation techniques in general and stated preference methods specifically are affected by uncertainty, stemming from gaps in knowledge about ecosystem dynamics, human preferences and technical issues in the valuation process. There is a need to include uncertainty issues in valuation studies and to acknowledge the limitations of valuation techniques in situations of radical uncertainty or ignorance about regime shifts.

- Valuation results will be heavily dependent on social, cultural and economic contexts, the boundaries of which may not overlap with the delineation of the relevant ecological system. Better valuation can be achieved by identifying and involving relevant stakeholders.

- Despite the difficulties of transferring valuation approaches and results between world regions, Benefits Transfer can be a practical, swift and cheap way to get an estimate of the value of local ecosystems, particularly when the aim is to assess a large number of diverse ecosystems. Values will vary with the characteristics of the ecosystem and the beneficiaries of the services it provides. Correcting values accordingly is advised when there are significant differences between the sites where the primary values are taken from and the sites to which values are to be transferred. Transfer errors are unavoidable and if highly precise estimates are needed, primary valuation studies should be commissioned.

- Monetary valuation can provide useful information about changes to welfare that will result from ecosystem management actions, but valuation techniques have limitations that are as yet unresolved. Valuation practioners should present their results as such, and policy makers should interpret and use valuation data accordingly.

- The limitations of monetary valuation are especially important as ecosystems approach critical thresholds and ecosystem change is irreversible or reversible only at prohivitive cost. Under conditions of high or radical uncertainty and existence of ecological thresholds, policy should be guided by the "safe-minimum-standard" and "precautionary approach" principles.

1 Introduction

Economics, as the study of how to allocate limited resources, relies on valuation to provide society with information about the relative level of resource scarcity. The value of ecosystem services and biodiversity is a reflection of what we, as a society, are willing to trade off to conserve these natural resources. Economic valuation of ecosystem services and biodiversity can make explicit to society in general and policy making in particular, that biodiversity and ecosystem services are scarce and that their depreciation or degradation has associated costs to society. If these costs are not imputed, then policy would be misguided and society would be worse off due to misallocation of resources.

Economically speaking, an asset is scarce if its use carries opportunity costs. That is, in order to obtain one additional unit of the good one must give up a certain amount of something else. In economic terms, quantifying and valuing ecosystem services are no different from quantifying and valuing goods or services produced by humans. In practice, however, valuing ecosystem services is problematic. There are reasonable estimates of the value of many provisioning services – in cases where well-developed markets exist – but there are few reliable estimates of the value of most non-marketed cultural and regulating services (Carpenter, 2006, Barbier et al., 2009). The problem is that since most ecosystem services and biodiversity are public goods, they tend to be overconsumed by society.

From an economic point of view, biodiversity (and ecosystems) can broadly be seen as part of our natural capital, and the flow of ecosystem services is the 'interest' on that capital that society receives (Costanza and Daly, 1992). Just as private investors choose a portfolio of capital to manage risky returns, we need to choose a level of biodiversity and natural capital that maintains future flows of ecosystem services in order to ensure enduring environmental quality and human well-being, including poverty alleviation (Perrings et al., 2006).

The basic assumption underlying the present chapter is that society can assign values to ecosystem services and biodiversity only to the extent that these fulfill needs or confer satisfaction to humans either directly or indirectly (although different forms of utilitarianism exist; see Goulder and Kennedy, 1997). This approach to valuing ecosystem services is based on the intensity of changes in people's preferences under small or marginal changes in the quantity or quality of goods or services. The economic conception of value is thus anthropocentric and for the most part instrumental in nature, in the sense that these values provide information that can guide policy making. This valuation approach, as discussed in chapter 4, should be used to complement, but not substitute other legitimate ethical or scientific reasoning and arguments relating to biodiversity conservation (see: Turner and Daily, 2008).

Valuation plays an important role in creating markets for the conservation of biodiversity and ecosystem services, for instance through Payments for Ecosystem Services (Engel et al., 2008; Pascual et al., 2010). Such market creation process requires three main stages: demonstration of values, appropriation of values and sharing the benefits from conservation (Kontoleon and Pascual, 2007). Demonstration refers to the identification and measurement of the flow of ecosystem services and their values (see also Chapters 2 and 3). Appropriation is the process of capturing some or all of the demonstrated and measured values of ecosystem services so as to provide incentives for their sustainable provision. This stage in essence ‘internalises’, through market systems, demonstrated values of ecosystem services so that those values affect biodiversity resource use decisions. Internalisation is achieved by *correcting* markets when they are ‘incomplete’ and/or *creating* markets when they are all-together missing. In the benefit sharing phase, appropriation mechanisms must be designed in such a manner that the captured ecosystem services benefits are distributed to those who bear the costs of conservation.

The concept of *total economic value* (TEV) of ecosystems and biodiversity is used thoughout this chapter. It is defined as the sum of the values of all service flows that natural capital generates both now and in the future – appropriately discounted. These service flows are valued for marginal changes in their provision. TEV encompasses all components of (dis)utility derived from ecosystem services using a common unit of account: money or any market-based unit of measurement that allows comparisons of the benefits of various goods. Since in many societies people are already familiar with money as a unit of account, expressing relative preferences in terms of money values may give useful information to policy-makers.

This chapter reviews the variety of taxonomies and classifications of the components of TEV and valuation tools that can be used to estimate such components for different types of ecosystem services. Given the complex nature of ecosystem services, economic valuation faces important challenges, including the existence of ecological thresholds and non-linearities, how to incorporate the notion of resilience of socio-ecological systems, the effects of uncertainty and scaling up estimated values of ecosystem services. This chapter reviews these challenges and from best practice provides guidelines for dealing with them when valuing ecosystems, ecosystem services and biodiversity.

An important note that should be kept in mind when reading this chapter is that while it follows the previous chapters in its conceptual approach to ecosystem services (see chapters 1 and 2), it also acknowledges that ecologists have multiple ways of framing and understanding ecosystems and that only some of these are compatible with a stock-flow model, or capital and interest analogy, of economics as it is presented here.

The chapter is structured as follows: Section 2 starts by asking the basic question of why we need to value ecosystem services and what types of values may be estimated that can have an effect in environmental decision-making, following the TEV approach.

In section 3, we look critically at the main methods used to estimate the various components of the TEV of ecosystem services and biodiversity. A summary and a brief description of each of these methods is provided, as well as a discussion of the appropriateness of using certain methods to value particular ecosystem services and value components. We also address various types of uncertainty inherent to valuation techniques.

Section 4 considers the insurance value of ecosystems by discussing related concepts such as resilience, option, quasi-option, and insurance value of biodiversity. Valuation results will vary along social, cultural and economic gradients and institutional scales will rarely correspond to the spatial scale of the relevant ecosystem and its services. Section 5 addresses these topics by covering stakeholder involvement, participatory valuation methods and the particular challenges of performing valuation studies in developing countries.

In section 6, we turn to benefits transfer, a widespreadly used technique to estimate values when doing primary studies is too costly in time or money. This section will present existing techniques for doing benefits transfer and discuss modifications needed to address problems that may arise when applying it across differing ecological, social and economic contexts. Section 7 concludes and reflects on the role of using value estimates to inform ecosystem policy.

2 Economic valuation of ecosystem services

It is difficult to agree on a philosophical basis for comparing the relative weights of intrinsic and instrumental values of nature. Box 1 presents briefly some of the main positions in this debate. Notwithstanding alternative views on valuation as discussed in chapter 4, this chapter sets the background and methods of economic valuation from the utilitarian perspective. Economic value refers to the value of an asset, which lies in its role in attaining human goals, be it spiritual enlightenment, aesthetic pleasure or the production of some marketed commodity (Barbier et al., 2009). Rather than being an inherent property of an asset such as a natural resource, value is attributed by economic agents through their willingness to pay for the services that flow from the asset. While this may be determined by the objective (e.g. physical or ecological) properties of the asset, the willingness to pay depends greatly on the socio-economic context in which valuation takes place – on human preferences, institutions, culture and so on (Pearce, 1993; Barbier et al., 2009).

Box 1: The intrinsic *versus* instrumental values controversy

Ethic and aesthetic values have so far constituted the core of the rationale behind modern environmentalism, and the recent incorporation of utilitarian arguments has opened an intense debate in the conservation community. Whereas ecologists have generally advocated biocentric perspectives based on intrinsic ecological values, economists adopt anthropocentric perspectives that focus on instrumental values. A main issue in this debate is the degree of complementarity or substitutability of these two different approaches when deciding on the conservation of biodiversity and ecosystem services. Some authors consider these two rationales to be complementary and see no conflict in their simultaneous use (e.g., Costanza, 2006). Others argue that adopting a utilitarian perspective may induce societal changes that could result in an instrumental conception of the human-nature relationship based increasingly on cost-benefit rationales (McAuley, 2006). Findings from behavioral experiments suggest that whereas some complementarity is possible, economic incentives may also undermine moral motivations for conservation (Bowles, 2008).

2.1 Why valuation?

One overarching question is why we need to value ecosystem services and biodiversity. Economics is about choice and every decision is preceded by a weighing of values among different alternatives (Bingham et al., 1995). Ecological life support systems underpin a wide variety of ecosystem services that are essential for economic performance and human well-being. Current markets, however, only shed information about the value of a small subset of ecosystem processes and components that are priced and incorporated in transactions as commodities or services. This poses structural limitations on the ability of markets to provide comprehensive pictures of the ecological values involved in decision processes (MA, 2005). Moreover, an information failure arises from the difficulty of quantifying most ecosystem services in terms that are comparable with services from human-made assets (Costanza et al., 1997). From this perspective, the logic behind ecosystem valuation is to unravel the complexities of socio-ecological relationships, make explicit how human decisions would affect ecosystem service values, and to express these value changes in units (e.g., monetary) that allow for their incorporation in public decision-making processes (Mooney et al., 2005).

Economic decision-making should be based on understanding the changes to economic welfare from small or marginal changes to ecosystems due to, e.g., the logging of trees in a forest or the restoration of a polluted pond (Turner et al., 2003). Value thus is a *marginal* concept insofar that it refers to the impact of small changes in the state of the world, and not the state of the world itself. In this regard, the value of ecological assets, like the value of other assets, is individual-based and subjective, context dependent, and state-dependent (Goulder and Kennedy, 1997, Nunes and van den Bergh, 2001). Estimates of economic value thus reflect only the current choice pattern of all human-made, financial and natural resources given a multitude of socio-ecological conditions such as preferences, the distribution of income and wealth, the state of the natural environment, production technologies, and expectations about the future (Barbier et al., 2009). A change in any of these variables affects the estimated economic value.

In summary, there are at least six reasons for conducting valuation studies:

- Missing markets
- Imperfect markets and market failures
- For some biodiversity goods and services, it is essential to understand and appreciate its alternatives and alternative uses.
- Uncertainty involving demand and supply of natural resources, especially in the future.
- Government may like to use the valuation as against the restricted, administered or operating market prices for designing biodiversity/ecosystem conservation programs
- In order to arrive at natural resource accounting, for methods such as Net Present Value methods, valuation is a must.

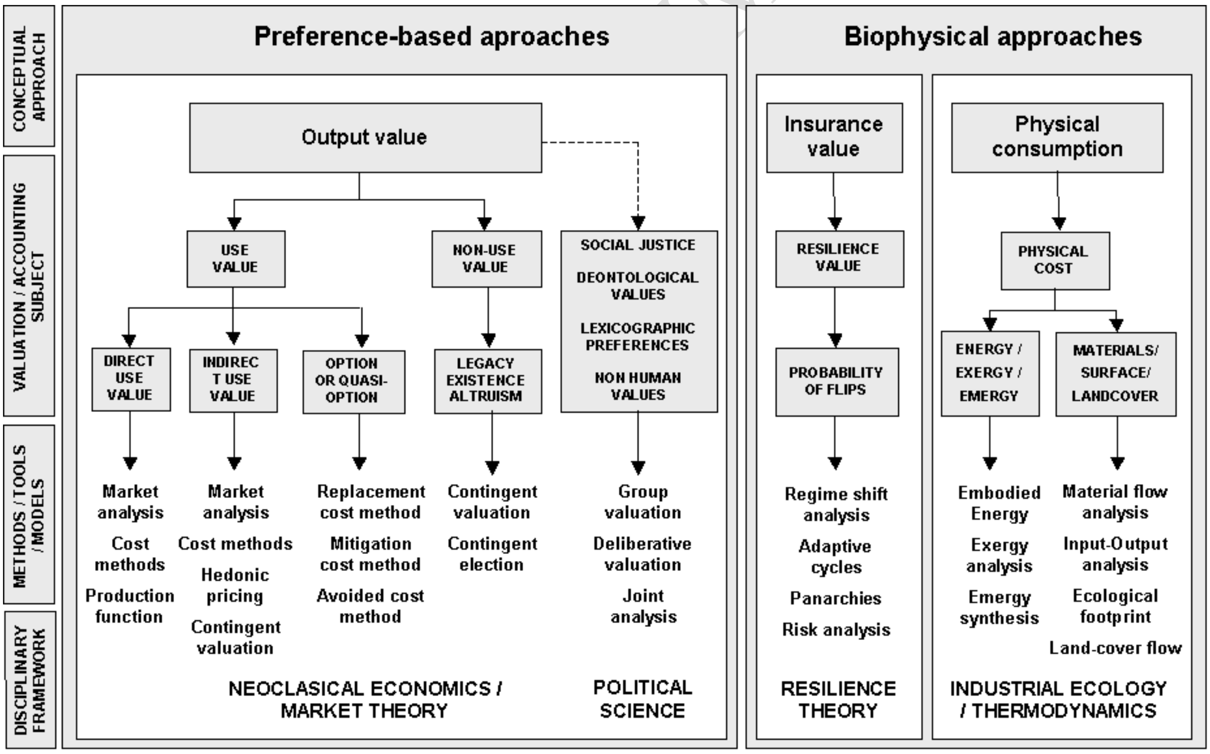
2.2 Valuation paradigms

Since there are multiple theories of value, valuation exercises should ideally, i) acknowledge the existence of alternative, often conflicting, valuation paradigms, and ii) be explicit about the valuation paradigm that is being used and its assumptions. A review on the approaches to valuation makes it possible to identify two well-differentiated paradigms for valuation: *biophysical* methods, constituted by a variety of biophysical approaches, and *preference-based* methods, which are more commonly used in economics. These methods are summarized in Figure 1:

Biophysical valuation uses a “cost of production" perspective that derives values from measurements of the physical costs (e.g., in terms of labor, surface requirements, energy or material inputs) of producing a given good or service. In valuing ecosystem services and biodiversity, this approach would consider the physical costs of maintaining a given ecological state. Box 2 provides a short discussion about biophysical approaches to valuation and accounting as an alternative to the dominant preference-based methods.

Box 2: Biophysical approaches to valuation and accounting

A number of economists have advocated biophysical measurements as a basis for valuation exercises. In contrast to preference-based approaches, biophysical valuation methods use a "cost of production" approach, as did some value theories in classical economics (e.g., the Ricardian and Marxist embodied labor theory of value). Biophysical approaches assess value based on the intrinsic properties of objects by measuring underlying physical parameters (see Patterson, 1998 for a review). Biophysical measures are generally more useful for the valuation of natural capital stocks than for valuation at the margin of flows of ecosystem services. This is particularly true when ecosystem services have no direct biophysical expression as in the case of some cultural services. In particular, biophysical measures can be especially useful for calculating depreciation of natural capital within a strong sustainability framework (which posits that no substitution is possible between human-made and natural resources). Examples of biophysical methods for the valuation or accounting of natural capital are embodied energy analysis (Costanza 1980), emergy analysis (Odum 1996), exergy analysis (Naredo, 2001; Valero et al., in press), ecological footprint (Wackernagel et al., 1999), material flow analysis (Daniels and Moore, 2002), land-cover flow (EEA, 2006), and Human Appropriation of Net Primary Production (HANPP) (Schandl et al., 2002).



Source: drafted from Gómez-Baggethun and de Groot, in press

Figure 1: Approaches for the estimation of nature's values.

In contrast to biophysical approaches to valuation, preference-based methods rely on models of human behavior and rest on the assumption that values arise from the subjective preferences of individuals. This perspective assumes that ecosystem values are commensurable in monetary terms, among themselves as well as with human-made and financial resources, and that subsequently, monetary measures offer a way of establishing the trade offs involved in alternative uses of ecosystems (for controversies on commensurability of value types see Box 3).

It should be noted that the biophysical and the preference-based approaches stem from different axiomatic frameworks and value theories, and therefore are not generally compatible. There is an ongoing debate about the need to use multiple units of measurement and notions of value in

Box 3: Conflicting valuation languages and commensurability of values

Controversies remain concerning the extent to which different types or dimensions of value can be reduced to a single rod of measure. Georgescu-Roegen (1979) criticized monism in applying theories of value, either preference-based or biophysical, as being a form of reductionism. Similarly, Martínez-Alier (2002) states that valuation of natural resources involves dealing with a variety of conflicting languages of valuation – e.g., economic, aesthetic, ecological, spiritual – that can not be reduced to a single rod of measure. This perspective emphasises "weak comparability" of values (O'Neill, 1993; Martínez-Alier et al., 1998) that puts values in a relation of "incommensurability" with each other. According to this view, decision support tools should allow for the integration of multiple incommensurable values. Multi-criteria analysis (MCA) makes possible the formal integration of multiple values after each of them has been assigned a relative weight (Munda, 2004). Like in monetary analysis, the output of MCA is a ranking of preferences that serve as a basis for taking decisions among different alternatives, but without the need to convert all values to a single unit (the result is an ordinal and not a cardinal ranking). MCA thus is a tool that accounts for complexity in decision-making processes. A weaknesses of this method is that the weighing of values can be easily biased by the scientists, or if the process is participatory, by power asymmetries among stakeholders. Transparent deliberative processes can reduce such risks, but also involve large amount of time and resources that are not generally available to decision makers (Gómez-Baggethun and de Groot, 2007).

environmental valuation (for brief overview of controversies on commensurability of value types see Box 3). This chapter deals primarily with *preference-based approaches*, and the terms economic valuation and monetary valuation are used interchangeably.

2.3 The TEV framework and value types

From an economic viewpoint, the value (or system value) of an ecosystem should account for two distinct aspects. The first is the aggregated value of the ecosystem service benefits provided in a given state, akin to the concept of TEV. The second aspect relates to the system's capacity to maintain these values in the face of variability and disturbance. The former has sometimes been referred to "output" value, and the latter has been named "insurance" value (Gren et al., 1994; Turner et al., 2003; Balmford et al., 2008) (Figure 2).

It should be emphasized that "total" in "total economic value" is summed across categories of values (i.e., use and non-use values) measured under marginal changes in the socio-ecological system, and

not over ecosystem or biodiversity (resource) units in a constant state. Recent contributions in the field of ecosystem services have stressed the need to focus on the end products (benefits) when valuing ecosystem services. This approach helps to avoid double counting of ecosystem functions, intermediate services and final services (Boyd and Banzhaf, 2007; Fisher et al., 2009).

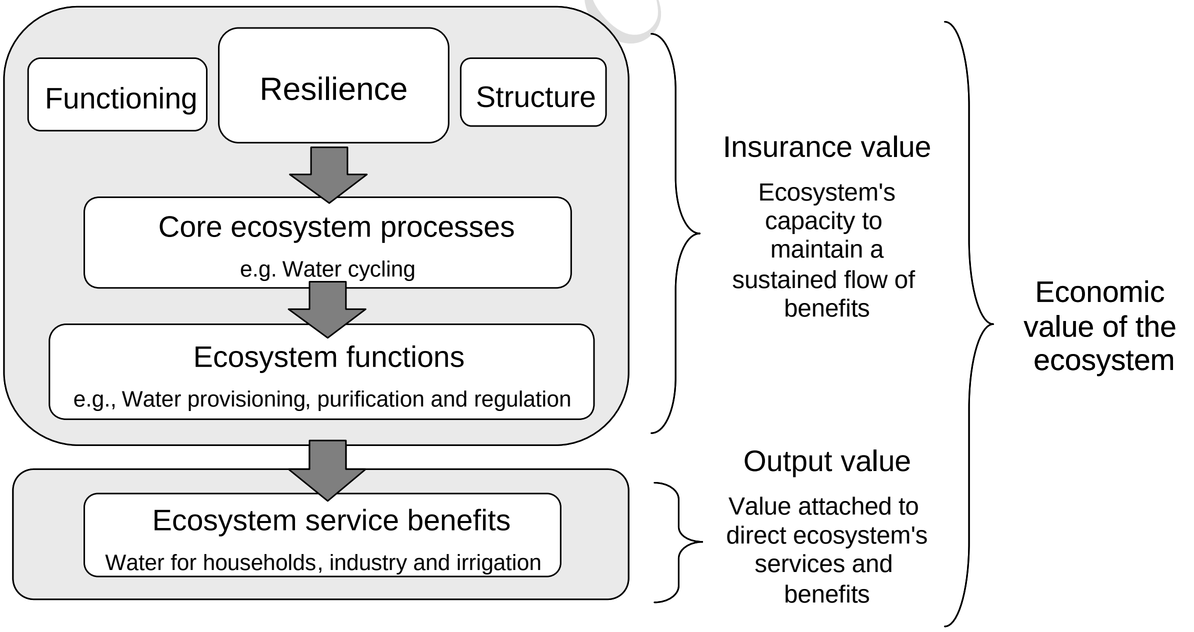


Figure 2: Insurance and output value as part of the economic value of the ecosystem
The figure poses *insurance value* (related to the ecosystem's resilience and *output value* (related to ecosystem service benefits) as the two main components of the economic value of the ecosystem.

The insurance value of ecosystems is closely related to the system's resilience and self-organizing capacity. The notion of resilience relate to the ecosystems' capacity to absorb shocks and reorganize so as to maintain its essential structure and functions, i.e., the capacity to remain at a given ecological state or avoid regime shifts (Holling, 1973; Walker et al., 2004). Securing ecosystem resilience involves maintaining minimum amounts of ecosystem infrastructure and processing capability that allows 'healthy' functioning. Such minimum ecological infrastructure can be approached through the concept of "critical natural capital" (Deutsch et al., 2003; Brand, 2009). The status of critical natural capital and related insurance values are sometimes recognized by the precautionary conservation of stocks, or setting safe minimum standards. However, the question remains how to measure resilience and critical natural capital in economic terms. These thorny issues are further discussed in more detail in section 4 of this chapter.

Benefits corresponding to the "output value" of the ecosystem can span from disparate values such as the control of water flows by tropical cloudy forests or the mitigation of damages from storms and other natural hazards by mangroves. The elicitation of these kinds of values can generally be handled with the available methods for monetary valuation based on direct markets, or, in their absence, on revealed or stated preferences techniques as will be discussed later.

Within the neoclassical economic paradigm, ecosystem services that are delivered and consumed in the absence of market transactions can be viewed as a form of positive externalities. Framing this as a market failure, the environmental economics literature has developed since the early 1960s a range of methods to value these "invisible" benefits from ecosystems, often with the aim of incorporating them into extended cost-benefit analysis and internalising the externalities. In order to comprehensively capture the economic value of the environment, different types of economic values neglected by markets have been identified, and measurements methods have been progressively refined. In fact, valuation of non-marketed environmental goods and services is associated with a large and still expanding literature in environmental economics.

Since the seminal work by Krutilla (1967), total (output) value of ecosystems has generally been divided into use- and non-use value categories, each subsequently disaggregated into different value components (Figure 3). A summary of the meaning of each component is provided in Table 1 based on Pearce and Turner (1991); de Groot et al. (2002), de Groot (2006) and Balmford et al. (2008).

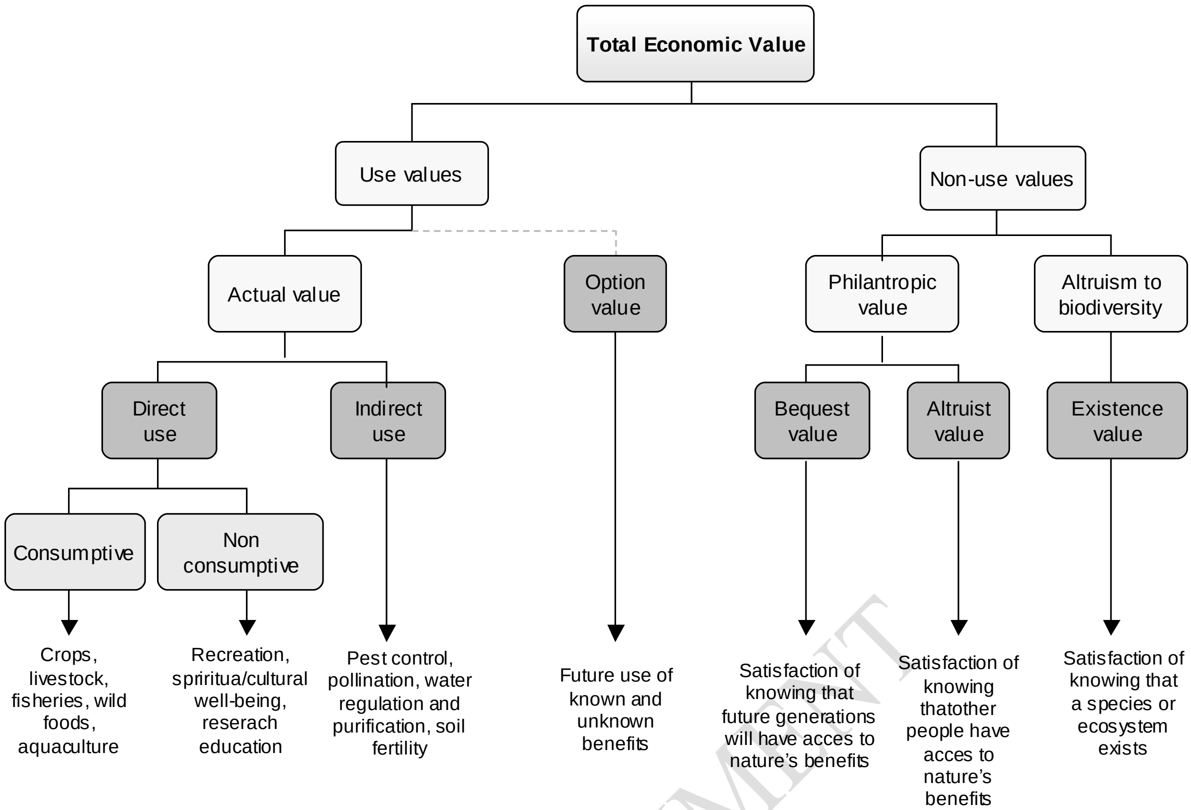


Figure 3: Value types within the TEV approach

Figure 3 reviews the value types that are addressed in the literature on nature valuation. Boxes in dark gray and the examples below the arrows are those that are directly addressed by value elicitation methods related to the TEV framework.

Table 1: A typology of values

Value type	Value sub-type	Meaning
Use values	Direct use value	Results from direct human use of biodiversity (consumptive or non consumptive).
	Indirect use value	Derived from the regulation services provided by species and ecosystems
	Option value	Relates to the importance that people give to the future availability of ecosystem services for personal benefit (option value in a strict sense).
Non-use values	Bequest value	Value attached by individuals to the fact that future generations will also have access to the benefits from species and ecosystems (intergenerational equity concerns).
	Altruist value	Value attached by individuals to the fact that other people of the present generation have access to the benefits provided by species and ecosystems (intragenerational equity concerns).
	Existence value	Value related to the satisfaction that individuals derive from the mere knowledge that species and ecosystems continue to exist.

Use values can be associated with private or quasi-private goods, for which market prices usually exist. Use values are sometimes divided further into two categories: (a) *Direct use value*, related to the benefits obtained from direct use of ecosystem service. Such use may be extractive, which entails consumption (for instance of food and raw materials), or non-extractive use (e.g., aesthetic benefits from landscapes). (b) *Indirect use values* are usually associated with regulating services, such as air quality regulation or erosion prevention, which can be seen as public services which are generally not reflected in market transactions.

Extending the temporal frame in which values are considered allows for the possibility of valuing the option of the future use of a given ecosystem service. This is often referred to as *option value* (Krutilla and Fisher, 1975). It is worth noting, however, that the consideration of option value as a true component of the TEV has been contested (Freeman, 1993). From this perspective, option value can be understood as a way of framing TEV under conditions of uncertainty, as an insurance premium or as the value of waiting for the resolution of uncertainty. In the latter case, it is generally known as *quasi-option value*.

An example to illustrate uncertainties surrounding the potential future uses and related option value of ecosystems is given by bioprospecting activities to discover potential medicinal uses of plants. Crucial issues in this example involve the question on whether or not any particular organism will prove to be of commercial use in the future; and what commercial uses will need to be developed over time. For a more extensive discussion, see section 4.

Non-use values from ecosystems are those values that do not involve direct or indirect uses of ecosystem service in question. They reflect satisfaction that individuals derive from the knowledge that biodiversity and ecosystem services are maintained and that other people have or will have access to them (Kolstad, 2000). In the first case, non-use values are usually referred to as *existence values*, while in the latter they are associated with *altruist values* (in relation to intra-generational equity concerns) or *bequest values* (when concerned with inter-generational equity).

It should be noted that non-use values involve greater challenges for valuation than do use values since non-use values are related to moral, religious or aesthetic properties, for which markets usually do not exist. This is different from other services which are associated with the production and valuation of tangible *things* or *conditions*. Cultural services and non-use values in general involve the production of *experiences* that occur in the valuer's mind. These services are therefore co-produced by ecosystems and people in a deeper sense than other services (Chan et al., in press). Table 2 provides an overview of the links between different categories of values of ecosystem services. The aggregation of these value categories is reflected in the TEV.

Table 2: **Valuing ecosystem services through the TEV framework**
N.A.= Non Applicable

Group	Service	Direct Use	Indirect use	Option value	Non-use value
Provisioning	Includes: food; fibre and fuel; biochemicals; natural medicines, pharmaceuticals; fresh water supply	*	NA	*	NA
Regulating	Includes: air-quality regulation; climate regulation; water regulation; natural hazard regulation, carbon storage, nutrient recycling, micro-climatic functions etc.	NA	*	*	NA
Cultural	Includes: cultural heritage; recreation and tourism; aesthetic values	*	NA	*	*
Habitat	Includes: primary production; nutrient cycling; soil formation	*Habitat services are valued through the other categories of ecosystem services*			

3 Valuation methods, welfare measures and uncertainty

3.1 Valuation methods under the TEV approach

Within the TEV framework, values are derived, if available, from information of individual behavior provided by market transactions relating directly to the ecosystem service. In the absence of such information, price information must be derived from parallel market transactions that are associated indirectly with the good to be valued. If both direct and indirect price information on ecosystem services are absent, hypothetical markets may be created in order to elicit values. These situations correspond to a common categorization of the available techniques used to value ecosystem services: (a) direct market valuation approaches, (b) revealed preference approaches and (c) stated preferences approaches (Chee, 2004). Below, a brief description of each method is provided together with a discussion on its strengths and weaknesses. We also discuss the adequacy of each method for different valuation conditions, purposes, ecosystem service types and value types to be estimated.

3.1.1 Direct market valuation approaches

Direct market valuation approaches are divided into three main approaches (a) market price-based approaches, (b) cost-based approaches, and (c) approaches based on production functions. The main advantage of using these approaches is that they use data from actual markets, and thus reflect actual preferences or costs to individuals. Moreover, such data – i.e. prices, quantities and costs- exist and thus are relatively easy to obtain.

Market price-based approaches are most often used to obtain the value of provisioning services, since the commodities produced by provisioning services are often sold on, e.g., agricultural markets. In well-functioning markets preferences and marginal cost of production are reflected in a market price, which implies that these can be taken as accurate information on the value of commodities. The price of a commodity times the marginal product of the ecosystem service is an indicator of the value of the service, consequently, market prices can also be good indicators of the value of the ecosystem service that is being studied.

Cost-based approaches are based on estimations of the costs that would be incurred if ecosystem service benefits needed to be recreated through artificial means (Garrod and Willis, 1999). Different techniques exist, including, (a) the *avoided cost method*, which relates to the costs that would have been incurred in the absence of ecosystem services, (b) *replacement cost method*, which estimates the costs incurred by replacing ecosystem services with artificial technologies, and (c) *mitigation* or *restoration cost method*, which refers to the cost of mitigating the effects caused by to the loss of ecosystem services or the cost of getting those services restored.

Production function-based approaches (PF) estimate how much a given ecosystem service (e.g., regulating service) contributes to the delivery of another service or commodity which is traded on an existing market. In other words, the PF approach is based on the contribution of ecosystem services to the enhancement of income or productivity (Mäler, 1994; Patanayak and Kramer, 2001). The idea thus is that any resulting "improvements in the resource base or environmental quality" as a result of enhanced ecosystem services, "lower costs and prices and increase the quantities of marketed goods, leading to increases in consumers' and perhaps producers' surpluses" (Freeman 2003, p. 259). The PF approach generally consists of the following two-step procedure (Barbier, 1994). The first step is to determine the physical effects of changes in a biological resource or ecosystem service on an economic activity. In the second step, the impact of these changes is valued in terms of the corresponding change in marketed output of the traded activity. A distinction should be made then between the gross value of output and the value of the marginal product of the input.

Hence, the PF approach generally uses scientific knowledge on cause-effect relationships between the ecosystem service(s) being valued and the output level of marketed commodities. It relates to objective measurements of biophysical parameters. As Barbier et al. (2009) note, for many habitats where there is sufficient scientific knowledge of how these link to specific ecological services that support or protect economic activities, it is possible to employ the production function approach to value these services.

Limitations of direct market valuation approaches

Direct market valuation approaches rely primarily on production or cost data, which are generally easier to obtain than the kinds of data needed to establish demand for ecosystem services (Ellis and Fisher, 1987). However, when applied to ecosystem service valuation, these approaches have important limitations. These are mainly due to ecosystem services not having markets or markets being distorted.

The direct problems that arise are two-fold. If markets do not exist either for the ecosystem service itself or for goods and services that are indirectly related, then the data needed for these approaches are not available. In case where markets do exist but are distorted, for instance because of a subsidy scheme (see TEEB D1) or because the market is not fully competitive, prices will not be a good reflection of preferences and marginal costs. Consequently, the estimated values of ecosystem services will be biased and will not provide reliable information to base policy decisions on.

Some direct market valuation approaches have specific problems. Barbier (2007) illustrates that the replacement cost method should be used with caution, especially under uncertainty. The PF approach has the additional problem that adequate data on and understanding of the cause-effect linkages between the ecosystem service being valued and the marketed commodity are often lacking (Daily et al., 2000; Spash, 2000). In other words, "production functions" of ecosystem services are rarely understood well enough to quantify how much of a service is produced, or how changes in ecosystem condition or function will translate into changes in the ecosystem services delivered (Daily et al., 1997). Furthermore, the interconnectivity and interdependencies of ecosystem services may increase the likelihood of double-counting ecosystem services (Barbier, 1994; Costanza and Folke, 1997).

3.1.2 Revealed preference approaches

Revealed preference techniques are based on the observation of individual choices in existing markets that are related to the ecosystem service that is subject of valuation. In this case it is said that economic agents "reveal" their preferences through their choices. The two main methods within this approach are:

(a) The *travel cost method* (TC), which is mostly relevant for determining recreational values related to biodiversity and ecosystem services. It is based on the rationale that recreational experiences are associated with a cost (direct expenses and opportunity costs of time). The value of a change in the quality or quantity of a recreational site (resulting from changes in biodiversity) can be inferred from estimating the demand function for visiting the site that is being studied (Bateman et al., 2002; Kontoleon and Pascual, 2007).

(b) The *hedonic pricing (HP)* approach utilizes information about the implicit demand for an environmental attribute of marketed commodities. For instance, houses or property in general consist of several attributes, some of which are environmental in nature, such as the proximity of a house to a forest or whether it has a view on a nice landscape. Hence, the value of a change in biodiversity or ecosystem services will be reflected in the change in the value of property (either built-up or land that is in a (semi-) natural state). By estimating a demand function for property, the analyst can infer the value of a change in the non-marketed environmental benefits generated by the environmental good.

The main steps for undertaking a revealed preference valuation study are:

1. Determining whether a surrogate market exists that is related to the environmental resource in question.
2. Selecting the appropriate method to be used (travel cost, hedonic pricing).
3. Collecting market data that can be used to estimate the demand function for the good traded in the surrogate market.
4. Inferring the value of a change in the quantity/quality of an environmental resource from the estimated demand function.
5. Aggregating values across relevant population.
6. Discounting values where appropriate.

Limitations of revealed preference approaches

In revealed preferences methods, market imperfections and policy failures can distort the estimated monetary value of ecosystem services. Scientists need good quality data on each transaction, large data sets, and complex statistical analysis. As a result, revealed preference approaches are expensive and time-consuming. Generally, these methods have the appeal of relying on actual/observed behavior but their main drawbacks are the inability to estimate non-use values and the dependence of the estimated values on the technical assumptions made on the relationship between the environmental good and the surrogate market good (Kontoleon and Pascual, 2007).

3.1.3 Stated preference approaches

Stated preference approaches simulate a market and demand for ecosystem services by means of surveys on hypothetical (policy-induced) changes in the provision of ecosystem services. Stated preference methods can be used to estimate both use and non-use values of ecosystems and/or when no surrogate market exists from which the value of ecosystems can be deduced. The main types of stated preference techniques are:

(a) *Contingent valuation method (CV)*: Uses questionnaires to ask people how much they would be willing to pay to increase or enhance the provision of an ecosystem service, or alternatively, how much they would be willing to accept for its loss or degradation.

(b) Choice modeling (CM): Attempts to model the decision process of an individual in a given context (Hanley and Wright, 1998; Philip and MacMillan, 2005). Individuals are faced with two or more alternatives with shared attributes of the services to be valued, but with different levels of attribute (one of the attributes being the money people would have to pay for the service).

(c) Group valuation: Combines stated preference techniques with elements of deliberative processes from political science (Spash, 2001; Wilson and Howarth, 2002), and are being increasingly used as a way to capture value types that may escape individual based surveys, such as value pluralism, incommensurability, non-human values, or social justice (Spash, 2008).

As pointed out by Kontoleon and Pascual (2007), the main difference between CV and CM is that CV studies usually present one option to respondents. This option is associated with some (varying across respondents) price-tag. Respondents are then asked to vote on whether they would be willing to support this option and pay the price or if they would support the status quo (and not pay the extra price).[i] The distinction between voting as a market agent versus voting as a citizen has important consequences for the interpretation of CV results (Blamey et al., 1995).

Box 4: Steps for undertaking a contingent valuation study (Kontoleon and Pascual, 2006)

1. Survey design

- Start with focus group sessions and consultations with stakeholders to define the good to be valued.
- Decide the nature of the market, i.e., determine the good being traded, the status quo, and the improvement or deterioration level of the good that will be valued.
- Determine the quantity and quality of information provided over the traded 'good', who will pay for it, and who will benefit from it.
- Set allocation of property rights (determines whether a willingness-to-pay (WTP) or a willingness-to-accept (WTA) scenario is presented).
- Determine credible scenario and payment vehicle (tax, donation, price).
- Choose elicitation method (e.g. dichotomous choice vs. open-ended elicitation method).

2. Survey implementation and sampling

- Interview implementation: on site or face-to-face, mail, telephone, internet, groups, consider inducements to increase the response rate.
- Interviewers: private companies, researchers themselves.
- Sampling: convenience sample, representative and stratified sample.

3. Calculate measures of welfare change

- Open-ended – simple mean or trimmed mean (with removed outliers; note that this is a contentious step).
- Dichotomous choice – estimate expected value of WTP or WTA.

4. Technical validation

- Most CV studies will attempt to validate responses by investigating respondents WTP (or WTA) bids by estimating a bid function

5. *Aggregation and discounting*

- Calculating total WTP from mean/median WTP over relevant population – for example by multiplying the sample mean WTP of visitors to a site by the total number of visitors per annum.
- Discount calculated values as appropriate.

6. *Study appraisal*

- Testing the validity and reliability of the estimates produced

In a CM study, respondents within the survey are given a choice between several options, each consisting of various attributes, one of which is either a price or subsidy. Respondents are then asked to consider all the options by balancing (trading off) the various attributes. Either of these techniques can be used to assess the TEV from a change in the quantity of biodiversity or ecosystem services. Though the CV method is less complicated to design and implement, the CM approach is more capable of providing value estimates for changes in specific characteristics (or attributes) of an environmental resource. Box 4 provides the steps for undertaking a CV study and Box 5 gives an example of a CM study that aimed to value biodiversity.

Box 5: Example of valuing changes in biodiversity using a choice modeling study

In a study by Christie et al. (2007) the value of alternative biodiversity conservation policies in the UK was estimated using the CM method. The study assessed the total value of biodiversity under of alternative conservation policies as well as the marginal value of a change in one of the attributes (or characteristics) of the policies. The policy characteristics explored were familiarity of species conserved, species rarity, habitat quality, and type of ecosystem services preserved. The policies would be funded by an annual tax. An example of the choice options presented to individuals is presented below.

	POLICY LEVEL 1	**POLICY LEVEL 2**	**DO NOTHING (Biodiversity degradation will continue)**
Familiar species of wildlife	Protect *rare* familiar species from further decline	Protect *both rare and common* familiar species from further decline	Continued decline in the populations of familiar species
Rare, unfamiliar species of wildlife	Slow down the rate of decline of rare, unfamiliar species.	Stop the decline and ensure the recovery of rare unfamiliar species	Continued decline in the populations of rare, unfamiliar species
Habitat quality	Habitat restoration, e.g. by better management of existing habitats	Habitat re-creation, e.g. by creating new habitat areas	Wildlife habitats will continue to be degraded and lost
Ecosystem process	Only ecosystem services that have a direct impact on humans, e.g. flood defence are restored.	*All* ecosystem services are restored	Continued decline in the functioning of ecosystem processes
Annual tax increase	£100	£260	No increase in your tax bill

Respondents had to choose between Policy 1, Policy 2 and the status quo (do nothing). Studies such as these can provide valuable information in an integrated assessment of the impacts of trade policies on biodiversity. Consider a change in EU farmer subsidisation policies which will have a likely impact on the agricultural landscape in the UK. The network of hedge-groves that exists in the UK country side and which hosts a significant amount of biodiversity and yields important biodiversity services will be affected by such a revised subsidisation policy. Using results from the aforementioned CE study, policy makers can obtain an approximation of the value of the loss in biodiversity that might come about from a change in the current hedge-grove network.

Group valuation approaches have been acknowledged as a way to tackle shortcomings of traditional monetary valuation methods (de Groot et al., 2006). Main methods within this approach are *Deliberative Monetary Valuation (DMV)*, which aims to express values for environmental change in monetary terms (Spash, 2007, 2008), and *Mediated Modeling*.

In the framework of stated preference methods, it is easy to obtain other important data types for the assessment of ecosystem services, such as stated perceptions, attitudinal scales, previous knowledge, etc. All of these pieces of information have been shown to be useful in understanding choices and preferences (Adamowicz, 2004). Stated preference methods could be a good approximation of the relative importance that stakeholders attach to different ecosystem services (Nunes, 2002; Martín-López et al., 2007; García-Llorente et al., 2008), and sometimes could reveal potential conflicts among stakeholders and among alternative management options (Nunes et al., 2008).

Limitations of stated prefence approaches

Stated preference techniques are often the only way to estimate non-use values. Concerning the understanding of the *objective of choice*, it is often asserted that the interview process 'assures' understanding of the object of choice, but the hypothetical nature of the market has raised numerous questions regarding the validity of the estimates (Kontoleon and Pascual, 2007). The major question is whether respondents' hypothetical answers correspond to their behavior if they were faced with costs in real life.

One of the main problems that have been flagged in the literature on stated preference methods is the divergence between willingness-to-pay (WTP) and willingness-to-accept (WTA) (Hanneman, 1991; Diamond, 1996). From a theoretical perspective, WTP and WTA should be similar in perfectly competitive private markets (Willing, 1976, Diamond 1996). However, several studies have demonstrated that for identical ecosystem services, WTA amounts systematically exceed WTP (Vatn and Bromley, 1994). This discrepancy may have several causes: faulty questionnaire design or interviewing technique; strategic behavior by respondents and psychological effects such as 'loss aversion' and the 'endowment effect' (Garrod and Willis, 1999).

Another important problem is the "embedding", "part-whole bias" or "insensitivity to scope" problem (Veisten, 2007). Kahneman (1986) was among the first to claim that respondents in a CV survey were insensitive to scope – he observed from a study that people were willing to pay the same amount to prevent the drop in fish populations in one small area of Ontario as in all Ontario (see also Kahneman and Knetsch , 1992; Boyle et al., 1994, 1998; Desvousges et al., 1993; Diamond and Hausman, 1994), Diamond et al., 1993; Svedsäter, 2000).

There is also a controversy on whether non-use values are commensurable in monetary terms (Martínez-Alier et al. 1998; Carson et al., 2001). The problem here is whether, for instance, the religious or bequest value that may be attributed to a forest can be considered within the same framework as the economic value of logging or recreation in that forest. Such an extreme range of values may not be equally relevant to all policy problems, but the issue has remained largely unresolved for now.

Furthermore, the application of stated preference methods to public goods that are complex and unfamiliar has been questioned on the grounds that respondents cannot give accurate responses as their preferences are not fully defined (Svedsäter 2003). Sometimes stated preference methods incorporate basic upfront information in questionnaires (e.g. García-Llorente et al., 2008; Tisdell and Wilson, 2006; Wilson and Tisdell, 2005). Christie et al. (2006) argue that valuation workshops that provide respondents with opportunities to discuss and reflect on their preferences help to overcome some of the potential cognitive and knowledge constraints associated with stated preference methods. Typically deliberative monetary valuation methods will provide upfront information to stakeholders as well. The bias in deliberative monetary valuation approaches is supposedly less than in individual CV studies (de Groot et al., 2006). Such methods may further reduce non-response rates and increase respondets' engagement.

3.1.4 Choosing and applying valuation methods: forests and wetlands

The main purpose of this section is to provide examples about how valuation methods have been applied to elicit different kinds of ecosystem values. Here we present results, summarized in tables, from an extensive literature review about the application of valuation techniques to estimate a variety of values, particularly in forests and wetlands. The information here presented may help valuation practitioners to choose the appropriate valuation method, according to the concerned values. This section is short in scope because noumerous previous publications have dealt already with techniques' classification and applications.

As discussed extensively elsewhere (NRC, 1997; 2004; Turner et al., 2004; Chee, 2004), some valuation methods are more appropriate than others for valuing particular ecosystem services and for the elicitation of specific value components. Table 3 shows the links between specific methods and value components.

Table 3: Relationship between valuation methods and value types

<table>
<tr><th colspan="2">Approach</th><th>Method</th><th>Value</th></tr>
<tr><td rowspan="6">Market valuation</td><td>Price-based</td><td>Market prices</td><td>Direct and indirect use</td></tr>
<tr><td rowspan="3">Cost-based</td><td>Avoided cost</td><td>Direct and indirect use</td></tr>
<tr><td>Replacement cost</td><td>Direct and indirect use</td></tr>
<tr><td>Mitigation / Restoration cost</td><td>Direct and indirect use</td></tr>
<tr><td rowspan="2">Production -based</td><td>Production function approach</td><td>Indirect use</td></tr>
<tr><td>Factor Income</td><td>Indirect use</td></tr>
<tr><td colspan="2" rowspan="2">Revealed preference</td><td>Travel cost method</td><td>Direct (indirect) use</td></tr>
<tr><td>Hedonic pricing</td><td>Direct and indirect use</td></tr>
<tr><td colspan="2" rowspan="4">Stated preference</td><td>Contingent Valuation</td><td>Use and non-use</td></tr>
<tr><td>Choice modelling/ Conjoint Analysis</td><td>Use and non-use</td></tr>
<tr><td>Contingent ranking</td><td>Use and non-use</td></tr>
<tr><td>Deliberative group valuation</td><td>Use and non-use</td></tr>
</table>

Table 4 provides insight into and comments on some of the potential applications of methods in ecosystem services valuation and their references in the literature.

Method			**Comment /example**	**References**
Market valuation	Market Price		Mainly applicable to the "goods" (e.g. fish) but also some cultural (e.g. recreation) and regulating services (e.g. pollination).	Brown et al. 1990; Kanazawa 1993
	Cost based	Avoided cost	The value of the flood control service can be derived from the estimated damage if flooding would occur.	Gunawardena & Rowan 2005; Ammour et al. 2000; Breaux et al. 1995; Gren 1993
		Replace-ment cost	The value of groundwater recharge can be estimated from the costs of obtaining water from another source (substitute costs).	
		Mitigation/ restoration costs	E.g. cost of preventive expenditures in absence of wetland service (e.g. flood barriers) or relocation.	
	Production function / factor income		How soil fertility improves crop yield and therefore the income of the farmers, and how water quality improve-ments increase commercial fisheries catch and thereby incomes of fishermen.	Pattanayak & Kramer 2001
Revealed preferences	Travel Cost Method		E.g. part of the recreational value of a site is reflected in the amount of time and money that people spend while traveling to the site.	Whitten & Bennet 2002; Martín-López et al. 2009b
	Hedonic Pricing Method		For example: clean air, presence of water and aesthetic views will increase the price of surrounding real estate.	Bolitzer & Netusil 2000; Garrod & Willis 1991
Simulated valuation	Contingent Valuation Method (CVM)		It is often the only way to estimate non-use values. For example, a survey questionnaire might ask respondents to express their willingness to increase the level of wa-ter quality in a stream, lake or river so that they might enjoy activities like swimming, boating, or fishing.	Wilson & Carpenter 2000; Martín-López et al. 2007
	Choice modelling		It can be applied through different methods, which include choice experiments, contingent ranking, contingent rating and pair comparison.	Hanley & Wright 1998; Lii et al. 2004; Philip & MacMillan 2005
	Group valuation		It allows addressing shortcomings of revealed pre-ference methods such as preference construction during the survey and lack of knowledge of respondents about what they are being asked to allocate values.	Wilson & Howarth 2002; Spash 2008

Table 4: Monetary Valuation Methods and values: examples from the literature
Source: Compiled after King & Mazotta (2001), Wilson & Carpenter (1999), de Groot et al. (2006).

Regulation services have been mainly valued through avoided cost, replacement and restoration costs, or contingent valuation; cultural services through travel cost (recreation, tourism or science), hedonic pricing (aesthetic information), or contingent valuation (spiritual benefits –i.e. existence value); and provisioning services through methods based on the production function approach and direct market valuation approach (Martín-López et al., 2009a).

Drawn from a review of 314 peer reviewed valuation case studies (see Annex for references), Tables 5-6 provide quantitative information on valuation approaches and specific valuation techniques that have been used for the estimatino of particular categories and types of ecosystem services. Table 7 and Figure 4 zoom into values of wetlands and forests, following a review of valuation studies in these biomes.

The tables in Annex A provide an extensive overview of the valuation literature regarding the use of valuation methods to estimate different types of economic values of ecosystem services. The review covers only wetlands and forests, two biomes for which most studies could be found. Annex A contains a summary of the ecosystem services provided by these biomes and the techniques applied to them, as well as a table to summarize this information according to the typology of values from Table 1.

Tables A1 (a, b) show benefits/value types within each major (a) wetland and(b) forest ecosystem services categories, i.e. provisioning, regulating, cultural and supportive services. It also identifies valuation approaches used to estimate economic values. Table A2 (a, b) provides a complementary view that associates the ecosystem services from these two biomes with valuation approaches. Table A3 associates the benefits/value types in wetlands (a) and forest (b) ecosystem services per type of value (across various use/non use values).

Valuation method	Cultural	Provisioning	Regulating	Supporting
Avoided cost	1	2	**26**	0
Benefits transfer	9	3	4	6
Bio-economic modelling	0	1	0	0
Choice modelling	16	4	7	17
Consumer surplus	1	0	0	0
Contingent ranking	1	2	0	0
Conversion cost	0	1	0	0
CVM	**26**	10	9	**33**
Damage cost	0	0	6	0
Factor income/Production function	1	**33**	9	0
Hedonic pricing	5	1	0	0
Market price	0	7	3	0
Mitigation cost	0	2	3	0
Net price method	0	1	0	0
Opportunity cost	1	17	1	6
Participatory valuation	2	3	3	0
Public investments	0	1	1	**28**
Replacement cost	2	3	**20**	11
Restoration cost	1	2	6	0
Substitute goods	0	4	0	0
Travel cost method	**32**	3	3	0
Grand Total	**100%**	**100%**	**100%**	**100%**

Table 5: **Use of different valuation methods for valuing ecosystem services in the valuation literature**

Type of valuation approach	Cultural	Provisioning	Regulating	Supporting
Benefits transfer	9	3	4	6
Cost based	5	27	**61**	17
Production based	1	**33**	9	0
Revealed preference	38	18	7	28
Stated preference	**46**	19	19	**50**
Grand Total	**100%**	**100%**	**100%**	**100%**

Table 6: **Valuation approaches used for valuing ecosystem services**

Note: The data pertains to valuation studies published in peer reviewed literature. The total numbers of valuation studies are 314. See annex for references.

[i] If a WTA scenario is involved a policy option is described to respondents as to be associated with a specific subsidy amount. Respondents have to decide if they would want to support the policy and receive the subsidy or support the status quo and not receive any subsidy.

	Forests				Forests Total	Wetlands				Wetlands Total	Grand Total
Row Labels	**Cultural**	**Provisioning**	**Regulating**	**Supporting**		**Cultural**	**Provisioning**	**Regulating**	**Supporting**		
Benefits transfer	**2**	**1**	**5**	**0**	**2**	**16**	**6**	**3**	**25**	**9**	**5**
Benefits transfer	2	1	5	0	2	16	6	3	25	9	5
Cost based	**2**	**30**	**69**	**14**	**30**	**9**	**24**	**52**	**25**	**25**	**28**
Avoided cost	0	2	33	0	8	2	2	16	0	5	7
Conversion cost	0	0	0	0	0	0	2	0	0	1	0
Damage cost	0	0	10	0	2	0	0	0	0	0	1
Mitigation cost	0	4	3	0	2	0	0	3	0	1	2
Opportunity cost	0	20	3	7	10	2	13	0	0	6	8
Replacement cost	0	2	18	7	6	4	4	23	25	9	7
Restoration cost	2	1	3	0	2	0	4	10	0	4	3
Production based	**2**	**30**	**8**	**0**	**16**	**0**	**39**	**10**	**0**	**18**	**17**
Bio-economic modelling	0	0	0	0	0	0	2	0	0	1	0
Factor income/Prod func	2	30	8	0	16	0	37	10	0	17	16
Revealed preference	**57**	**27**	**13**	**36**	**32**	**20**	**4**	**0**	**0**	**8**	**22**
Consumer surplus	0	0	0	0	0	2	0	0	0	1	0
Hedonic pricing	7	2	0	0	3	4	0	0	0	1	2
Market price	0	12	5	0	7	0	0	0	0	0	4
Net price method	0	1	0	0	1	0	0	0	0	0	0
Public investments	0	0	3	36	3	0	4	0	0	1	3
Substitute goods	0	6	0	0	3	0	0	0	0	0	2
Travel cost method	50	5	5	0	16	13	0	0	0	4	11
Stated preference	**37**	**12**	**5**	**50**	**20**	**56**	**28**	**35**	**50**	**40**	**28**
Choice modelling	11	0	0	14	4	22	9	16	25	16	9
Contingent ranking	2	2	0	0	2	0	2	0	0	1	1
CVM	22	9	5	36	13	31	11	13	25	19	16
Participatory valuation	2	1	0	0	1	2	6	6	0	4	3
Grand Total	**100%**	**100%**	**100%**	**100%**	**100%**	**100%**	**100%**	**100%**	**100%**	**100%**	**100%**

Table 7: Proportion of valuation methods applied across ecosystem services regarding forests and wetlands, based on reviewed literature (see annex for references).

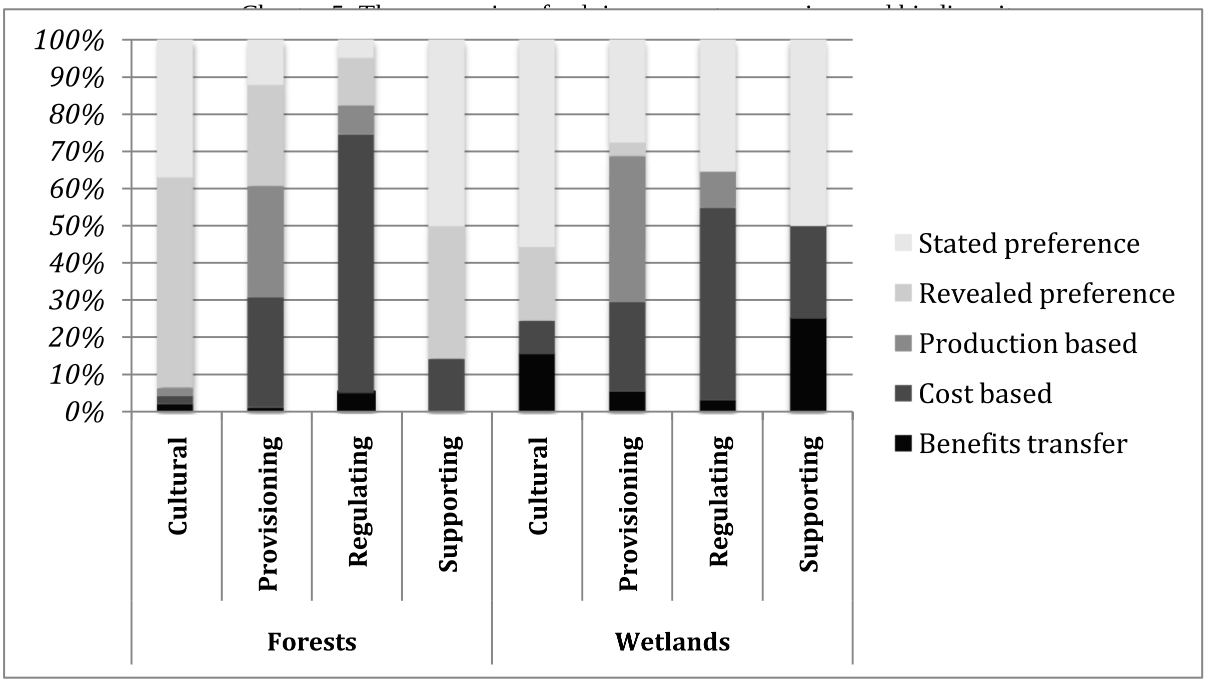


Figure 4: **Vauation approaches that have been used to value ecosystem services provided by forests and wetlands**

In sum, each of the methods explained herewith has its own strengths and shortcomings (Hanley and Spash, 1993; Pearce and Moran, 1994), and each can be particularly suitable for specific ecosystem services and value types. Table 8 summarizes the advantages and disadvantages of different techniques using the case of wetlands, but the information can also be used for other biomes.

Lastly, it should also be mentioned that there are "hybrid" valuation methods that can also be considered. For instance, it is theoretically possible to link a production function approach to stated preference method to estimate the economic value of, e.g., cultural services offered by totemic species. Allen and Loomis (2006) use such an approach to derive the value of species at lower trophic levels from the results of surveys of willingness to pay for the conservation of species at higher trophic levels. Specifically, they derive the implicit WTP for the conservation of prey species from direct estimates of WTP for top predators.

Valuation Technique	Advantage	Disadvantages
Market prices method. Use prevailing prices for goods and services traded in domestic or international.	Market prices reflect the private willingness to pay for wetland costs and benefits that are traded (e.g., fish, timber, fuelwood, recreation). They may be used to construct financial accounts to compare alternative wetland uses from the perspective of the individual or company con-cerned with private profit and losses. Price data are relatively easy to obtain.	Market imperfections and/or policy failures may distort market prices, which will therefore fail to reflect the economic value of goods or services to society as a whole. Seasonal variations and other effects on prices need to be considered when market prices are used in economic analysis.
Efficiency (shadow) prices method. Use of market prices but adjusted for transfer payments, market imperfections and policy distortions. May also incorporate distribution weights, where equa-lity concerns are made explicit. Shadow prices may also be calcu-lated for non-marketed goods.	Efficiency prices reflect the true economic value or opportunity cost, to society as a whole, of goods and services that are traded in domestic or international markets (e.g., fish, fuelwood, peat).	Derivation of efficiency prices is complex and may require substantial data. Decision-makers may not accept ‘artificial’ prices.
Hedonic pricing method. The value of an environmental ame-nity (such as a view) is obtained from property or labor markets. The basic assumption is that the observed property value (or wage) reflects a stream or benefits (or working conditions) and that it is possible to isolate the value of the relevant environmental amenity or attribute.	Hedonic pricing has the potential to value certain wetland functions (e.g., storm protection, groundwater recharge) in terms of their impact on land values, assuming that the wetland functions are fully reflected in land prices.	Application of hedonic pricing to the environmental functions of wetlands requires that these values are reflected in surrogate markets. The approach may be limited where markets are distorted, choices are constrained by income, information about environ-menttal conditions is not widespread and data are scarce.
Travel cost approach. The travel cost approach derives willingness to pay for environmental benefits at a specific location by using information on the amount of money and time that people spend to visit the location.	Widely used to estimate the value of recreational sites including public parks and wildlife services in developed countries. It could be used to estimate willingness to pay for eco-tourism to tropical wetlands in some developing countries.	Data intensive; restrictive assumptions about consumer behavior (e.g. multifunctional trips); results highly sensitive to statistical methods used to specify the demand relation-ship.
Production function approach. Estimates the value of a non-marketed resource or ecological function in terms of changes in economic activity by modeling the physical contribution of the resource or function to economic output.	Widely used to estimate the impact of wetlands and reef destruction, deforestation and water pollution, etc., on productive activities such as fishing, hunting and farming.	Requires explicit modeling of the ‘dose-response’ relationship be-tween the resources and some eco-nomic output. Application of the approach is most straightforward in the case of single use systems but becomes more complicated with multiple use systems. Problems may arise from multi-specification of the ecological-economic relationship or double counting.

Valuation Technique	Advantage	Disadvantages
Constructed market techniques. Measure of willingness to pay by directly eliciting consumer preferences.	Directly estimates Hicksian welfare measure – provides best theoretical measure of willingness to pay.	Practical limitations of constructed market techniques may detract from theoretical advantages, leading to poor estimates of true willingness to pay.
Simulated market (SM) constructs an experimental market in which money actually changes hands.	Controlled experimental setting permits close study of factors determining preferences.	Sophisticated decision and implementation may limit application in developing countries.
Contingent valuation methods (CVM) construct a hypothetical market to elicit respondents' willingness to pay.	Only method that can measure option and existence values and provide a true measure of total economic value.	Results sensitive to numerous sources of bias in survey design and implementation.
Contingent ranking (CR) ranks and scores relative preferences for amenities in quantitative rather than monetary terms.	Generates value estimate for a range of products and services without having to elicit willingness to pay for each.	Does not elicit willingness to pay directly, hence lacks theoretical advantages of other approaches. Being qualitative, can not be used directlyin policies (say for fixing cess, taxes etc.)
Cost-based valuation. Based on assumption that the cost of maintaining an environmental benefit is a reasonable estimate of its value. To estimate willingness to pay:	It is easier to measure the costs of producing benefits than the benefits themselves, when goods, services and benefits are non-marked. Approaches are less data and resource-intensive.	These second- best approaches assume that expenditure provides positive benefits and net benefits generated by expenditure match the original level of benefits. Even when these conditions are met, costs are usually not an accurate measure of benefits. So long as it's not clear whether it's worth it to replace a lost of damaged asset, the cost of doing so is an inadequate measure of damage.
Restoration cost (RSC) method uses costs of restoring ecosystem goods or services.	Potentially useful in valuing particular environmental functions.	Diminishing returns and difficulty of restoring previous ecosystem conditions make application of RSC questionable.
Replacement cost (RPC) method uses cost of artificial substitutes for environmental goods or services.	Useful in estimating indirect use benefits when ecological data are not available for estimating damage functions with first-best methods.	Difficult to ensure that net benefits of the replacement do not exceed those of the original function. May overstate willingness to pay if only physical indicators of benefits are available.
Relocation cost (RLC) method uses costs of relocating threatened communities.	Only useful in valuing environmental amenities in the face of mass dislocation such as a dam project and establishment of protected areas.	In practice, benefits provided by the new location are unlikely to match those of the original location.
Preventive expenditure (PE) approach uses the costs of preventing damage or degradation of environmental benefits.	Useful in estimating indirect use benefits with prevention technologies	Mismatching the benefits of investment in prevention to the original level of benefits may lead to spurious estimates of willingness to pay.

Valuation Technique	Advantage	Disadvantages
Damage costs avoided (D) approach relies on the assumption that damage estimates are a measure of value. It is not a cost-based approach as it relies on the use of valuation methods described above.	Precautionary principle applied here	Data or resource limitations may rule out first-best valuation methods.

Table 8: **Valuation techniques as applied to wetland studies**
(Source: Barbier et al. 1997).

3.2. Acknowledging uncertainty in valuation

In addition to the issues discussed in previous sections, uncertainty is another critical issue in the valuation of ecosystem services and biodiversity. This section addresses the role of uncertainty by reviewing the state of the art in the valuation literature. To do so, it is useful to distinguish between risk and uncertainty. Risk is associated with a situation where the possible consequences of a decision can be completely enumerated in terms of states of nature and probabilities assigned to each possibility (Knight, 1921 in Perman et al, 2003). In a Knightian sense, uncertainty is understood as the situation where the possible consequences of a decision can be fully enumerated but where a decision maker cannot assign probabilities objectively to these states. In addition, there is a more profound type of uncertainty where the decision maker cannot enumerate all of the possible consequences of a decision. This is usually referred to as 'radical uncertainty' or 'ignorance' (Perman et al 2003) and should be acknowledged when science cannot explain some complex functioning of ecosystems and biodiversity.[ii] In this chapter the term 'uncertainty' will refer to the one commonly used in economic valuation of the environment, i.e., the conflated risk and uncertainty notion as in Freeman (1993), unless the term "radical uncertainty" or "ignorance" is used instead.

Further, it is useful to distinguish three sources of uncertainty and radical uncertainty/ignorance. First, we may face uncertainty or/and ignorance in terms of the nature of the ecosystem services to be valued. Second, we may be uncertain or/and ignorant about the way people form their preferences about ecosystem services, i.e., the way they subjectively value changes in the delivery of ecosystem services and biodiversity. Lastly, another layer of uncertainty exists regarding the application of valuation tools. This is acknowledged here as technical uncertainty. In the following sections, these terms will be discussed where relevant, and best practice solutions discussed.

3.2.1 Uncertainty regarding the supply of ecosystem services

Beyond the problem of assigning probability distributions, radical uncertainty has tremendous implications for valuing biodiversity and ecosystem services. Science is starting to shed light about the role of biodiversity in terms of the delivery of supporting services, and robust information is still lacking on how biodiversity contributes to the ecological functions that translate into tangible benefits for society. For example, forested riparian corridors in agricultural landscapes clearly improve water quality and reduce sediment loads from upstream erosion, but ecologists have only a limited understanding of how species richness in riparian zones contribute to these ecosystem services (Jackson et al., 2007). In the same light, it is not straightforward to assign values to the services attributable to the diversity of tree species, rather than the stock of tree biomass or to the ecosystem as a whole. Usually valuation studies using stated preference methods rather than focusing on direct evidence about the link between 'biodiversity' (e.g. tree diversity) and peoples' preferences about such diversity, have mostly focused on more easily identifiable biological 'resources' or stocks (e.g. forests, wetlands and charismatic species) (Nunes and van den Bergh, 2001).

Beyond the more challenging effect of radical uncertainty, in cases where states of nature are identifiable and probability distributions can be objectively assigned by researchers, it is possible to resort to the use of *expected values* for those variables whose precise values cannot be known in advance. In this way uncertainty is dealt by weighting each potential outcome by the probability of its occurrence. In this case, we are dealing with the more palatable notion of Knightian risk, which is conflated with the standard notion of uncertainty in economic valuation. In this case, the valuation of a change in ecosystem services is based on the weighted outcomes of alternative states of the world. For example, a set of forest tree species, could be associated with an *expected* level of carbon capture given various rainfall patterns (states of nature). If probabilities can be assigned to these rainfall patterns, the amount of carbon that the forest can be expected to capture can be estimated by summing up the probability-weighted capture outcomes. Then, what is valued is the expected change in carbon capture associated with tree diversity given an objectively assigned probability distribution to rainfall patterns.

Examples from the literature dealing with ecosystem service valuation under uncertainty include the flow regulation in rivers and surge protection in coastal ecosystems which are fundamentally probabilistic. A promising approach is based on the expected damage function (EDF), akin to a dose-response approach but based on methodologies used in risk analysis. Barbier (2007) applies the EDF approach to value the storm protection service provided by a coastal wetland. The underlying assumption is that changes in wetland area affect the probability and severity of economically damaging storm events (states of nature) in coastal areas. More generally, this approach measures the WTP by measuring the total expected damages resulting from changes in ecosystem stocks. This approach has been used routinely in risk analysis and health economics (e.g., Barbier et al., 2009).

In the case of the coastal wetland example provided by Barbier (2007), a key piece of information becomes critical for estimating the value of wetlands in the face of economically damaging natural disasters: the influence of wetland area on the expected incidence of storm events. Provided that there is sufficient data on the incidence of past natural disasters and changes in wetland area in coastal regions, the first component can be dealt with by employing a *count data model* to estimate whether a change in the area of coastal wetlands, reduces the expected incidence of economically damaging storm events. Once the damage cost per event is known, the count data model yields the information to be used to calculate the value of wetlands in terms of protection against natural disasters.

The uncertainty of supply of ecosystem services makes stated preference methods significantly complex. This may be the reason why there are few examples where CV has considered valuation under uncertainty. In a seminal study, Brookshire et al. (1983) showed how option prices change when the uncertainty of supply (based on probabilistic risk) is reduced. Their WTP bid schedules were estimated by asking hunters their WTP given different probabilistic scenarios of the supply of threatened species such as grizzly bears and bighorn sheep in Wyoming. In another early application of CV under uncertainty, Crocker and Shogren (1991) valued landscape visibility changes under different accessibility contingencies of the sample of individuals being surveyed.. Their approach was based on eliciting the individuals' subjective perceptions about the probabilities of alternative landscape visibility states.

Generally, CV studies have resorted to measure respondents' risk perceptions, especially using so-called 'risk indexes' in order to obtain information about whether respondents feel concerned when considering an uncertain issue. Risk indexes, reflect individual beliefs about subjective probabilities of a given event occurring (e.g., the loss of a given species). In another CV application, Rekola and Pouta (2005), measure the value of forest amenities in Finland under uncertainty regarding forest regeneration cuttings. In this study, respondents' risk perceptions are measured and used to calculate the probability density function of expectations. They conclude that surveyed individuals may answer questions about risk perception inconsistently as people have a tendency to overestimate small probabilities, especially when these probabilities are connected with unwanted outcomes. The reason is that individuals may confound the subjective probability of the event occurring with the subjective perception about the severity of the event being perceived (e.g., the *feelings* about the loss of the species). This may undermine the use of risk indexes to use a probabilic approach within CV (see: Poe and Bishop, 1999; Rekola, 2004). This is a reason why stated preference practitioners tend to avoid using quantitative information about probabilities of provision of ecosystem services. Such information can undermine the studies in a way similar to how incentive compatible revelation of preferences can affect results (e.g., Carson and Groves. 2007).

3.2.2 Uncertainty with regard to preferences about ecosystem services

Valuation studies often assume that respondents know their preferences with certainty, i.e. they are aware how much they would be willing to pay for such ecosystem service provision. Empirical evidence in the stated preference literature suggests, however, that respondents are uncertain about their responses (Ready et al., 1995; Champ et al., 1997; Alberini et al., 2003; Akter et al., 2008). This is mainly due to respondents using a heuristic mode when processing information provided in one of several contingent valuation formats (e.g., interview, email), which tends to dominate over more systematic ways of information processing for decision-making (Bateman et al., 2004). This is compounded by an unfamiliar hypothetical nature of the market being recreated for sometimes unfamiliar or intangible goods such as the protection of a rare bird species in an unfamiliar location (Champ and Bishop, 2001; Schunn et al., 2000; Bateman, 2004).

Often an *ad hoc* way of dealing with preference uncertainty is to assume that people are expected-utility maximizers. This assumption makes it possible to calculate point estimates of expected willingness-to-pay for changes in ecosystem services. These calculations require that a random variable is added to individuals' utility functions, since arguably they do not know their true WTP for the service with certainty (Hanemann et al., 1996). Instead, they perceive that the true value of the service lies within an interval. A similar approach proposes that the level of individual preference uncertainty is determined by the magnitude of difference between a deterministic and a stochastic part of an individual's utility difference function (Loomis and Ekstrand, 1998).

There is no consensus about which method is more appropriate for measuring preference uncertainty in stated preference methods.[iii] There are three main approaches to deal with this kind of uncertainty in CVM. One is to request respondents to state how certain they are about their answer to the WTP question (e.g., Loomis and Ekstrand, 1998). Another one is to introduce uncertainty directly using multiple bounded WTP questions or a polychotomous choice model (e.g., Alberini et al., 2003). The third option is to request respondents to report a range of values rather than a specific value for the change in the provision of an ecosystem service (e.g., Hanley et al., 2009).

The first approach to deal with preference uncertainty in stated preference methods is the most straightforward one but one which does not solve the problem of uncertainty *per se*. It tries to uncover whether individuals' perceptions and attitudes to the good or service being valued are correlated with self-reported 'certainty scores'. The literature suggests some positive association between certainty scores and respondents' prior knowledge about the particular good being valued or respondents' attitudes towards the hypothetical market being confronted with (Loomis and Ekstrand, 1998).[iv]

The second approach introduces uncertainty directly into the WTP question by including uncertainty options. The idea is to include multiple bids in discrete choices by displaying a panel to respondents with suggested costs (WTP) on the rows and categories of certainty (e.g., from "extremely unlikely" to "extremely likely"), of whether respondents would be WTP the cost in exchange for a good or service in the columns (e.g., Alberini et al. 2003, Akter et al., forthcoming). The advantage of this approach is that it is possible to model the ordered structure of the data and identify threshold values, showing at which average bid levels people switch from one uncertainty level to another (Broberg, 2007). However, similar to the problems of using responses to uncertainty questions to re-classify WTP statements in stated preference methods, this polychotomous choice approach suffers from not knowing how respondents interpret concepts such as "very unlikely" and whether all respondents do so in the same way. [v]

The third approach is a promising alternative to the previous two approaches when people may prefer reporting a range of values rather than a specific value for the change in the provision of an ecosystem service. Hanley et al. (2009) suggest using a payment ladder to elicit peoples' WTP for changes in ecosystem services. In their example they value improvements in coastal water quality in Scotland and show that when using value ranges uncertainty is inversely related to the level of knowledge and experience with the good, although this effect only appears once a certain minimal level of experience has been acquired.

From the three approaches described above, the third approach appears to be the most promising for dealing with preference uncertainty. One issue that remains open though is the range of values to be used in this elicitation method. In addition, it is important to note that valuing an ecosystem service using the method presented in Hanley et al. (2009) only deals with one aspect of uncertainty about preferences as ecosystem services relevant to local respondents may not match with scientifically described ecosystem functions (Barkman et al., 2008).

3.2.3 Technical uncertainty due to applications of valuation tools

When deciding which valuation tools to use one should also think of the several conceptual, methodological and technical shortcomings associated with all valuation methods which add some further uncertainty to the estimated values. An extensive review of these issues is provided in Kontoleon et al. (2002). For the purposes of technical uncertainty that should be acknowledged in TEEB, two sets of issues must be noted: the first concerns the accuracy of valuation estimates and the second concerns the issue of discounting future values. Next we address the problem of the accuracy of valuation estimates elicited using standard valuation approaches and chapter 5 deals with the effect of different discount rates on the range of values that are estimated.

Measurement issues concern at least two key aspects of the problems concerning the accuracy of stated preference studies. One aspect is the *credibility* of the stated preferences. It is usually assumed that when using stated preference methods such as CV that respondents answer questions truthfully given the hypothetical nature of the technique. This issue is treated as a debate revolving around whether an upward "hypothetical bias" (the difference between purely hypothetical and actual statements of value) permeates CV estimates. Interestingly, a meta-analysis based on estimates from CV surveys to estimates with their counterparts based on revealed behavior techniques found no statistically significant upward hypothetical bias of CV methods (Carson et al., 1996). However the question remains whether estimates of non-use values elicited through stated preference methods are credible as there is no other approach to directly compare these values.

The second question is whether respondents answer truthfully only when it is in their interest to do so. While this problem is consistent with standard economic theory, this also means that responses depend critically on how well the surveys create incentives for the truthful revelation of preferences (Carson et al., 2008). For example, if an individual wishes to skew the results of the exercise, surveys do not generally include any explicit in-built incentive or mechanism that will constrain this sort of behaviour.,Hence the credibility of the results of a survey is a function of the quality of the survey design. The other problem of accuracy concerns the margin of error surrounding the valuation. This error will depend to some extent on the size of the sample and the nature of the good being valued, but it will necessarily remain fairly large and uncertain on account of the technique that is used.

As it has been mentioned in section 2, it should also be noted that a particularly prevalent error is the general use of WTP-type questions instead of WTA-type ones in stated preference surveys specially when the property rights of the goods or services being valued would warrant the WTA questions. This is so in spite of a sizeable literature establishing the presence of 'endowment effects' (Knetsch, 2005). Careful experiments reveal that even for market goods (e.g. coffee mugs, pens or candybars), WTA typically exceeds WTP (Kahneman et al., 1990). Further there is evidence that stated preference-based studies exhibit a rather substantial divergence between WTP and WTA results. A meta-analysis of 45 studies has found over a seven-fold difference between the two measures, on average (Horowitz and McConnell, 2002). Theoretical arguments against such disparities still are a matter of concern for valuation practitioners. It also provides ammunition against the use of stated preference methods and is taken as evidence that the CVM is a flawed valuation approach as it is inconsistent with neoclassical consumer theory in general and with its ability to measure consumer preferences (e.g., Diamond 1996, Hausman, 1993). Against these notorious criticisms, Practitioners of the CVM (e.g, Mitchell and Carson 1989) or the members of the NOAA panel (1993) recommend to use the WTP format for practical studies.[vi] Their reason is that since WTP generally turns out to be smaller than WTA, this is consistent with applying a 'conservative choice' to be on the safe side (NOAA 1993). But in this recommendation one may interpret some resignation with respect to the significance of CV results.

Accuracy problems also affect revealed preference and pricing techniques. The first problem has to do with the *availability* of revealed preference and market data that is required to undertake such valuation studies. Market data availability is about both quantity and quality of the data especially in the developing world where market data may suffer from poor quality that misrepresents reality. The second aspect of the accuracy of revealed preference and pricing techniques has to do with the fact that these methods (by their design) *cannot account for non-use values*. Hence, market data can only provide a lower bound estimate of the value of a change in biodiversity or ecosystem services.

In sum, valuation studies using various techniques can suffer from technical uncertainty due to accuracy problems or biases, examples being: i) the potential, e.g., hypothetical or strategic, biases that arise from the design of questionnaires in stated preference methods (Bateman et al., 2002), ii) the effect of assigning probabilistic scenarios in production function based approaches and iii) the influence of unstable market prices of substitutes or complements to natural resources in revealed preference methods (e.g., travel cost approach).

3.2.4 Data enrichment models and preference calibration as the way forward

One practical way to deal with at least two of the sources of uncertainty, namely technical uncertainty and to a lesser extent preference uncertainty is the use of the data enrichment or "data fusion" approach. The idea is to combine revealed and stated preference methods when valuing a given ecosystem service which is at least associated with clear direct use values. While this approach is not dominant in the valuation literature there are increasing calls from previous studies which have combined data and models to increase the reliability of the valuation estimates, for example to derive values for recreation, environmental amenity, cultural heritage and agrobiodiversity (e.g., Cameron, 1992, Adamowicz et al. 1994, Earnhart, 2001; Haab and McConnell, 2002; Birol et al., 2006). The main advantage of the data enrichment approach involving the combination of revealed and stated preference methods is that it overcomes two of the main problems associated with each of the two methods.

On the one hand while the advantage of using revealed preference methods is that it has a high "face validity" because the data reflect real choices and take into account various constraints on individual decisions, such as market imperfections, budgets and time (Louviere et al., 2000), it also suffers on the grounds that the new policy situation (after the change in the quality or the quantity of ecosystem services) may be outside the current set of experiences, i.e., outside the data range. Therefore, simulation of the new situation would involve extrapolation of available data outside the range used when estimating the model. In this case, combining information about the actual behavioural history of individuals with hypothetical changes to their behaviour through stated preference methods is seen as an obvious advantage of data fusion.

On the other hand, the purely hypothetical aspect of the latter can be checked against actual behaviour through revealed preference methods. Using revealed preference data assures that estimation is based on observed behaviour and combining it with stated preference responses to hypothetical changes of ecosystem services allows the identification of value ranges that otherwise would not be identified. This way, the amount of information increases, and findings can be cross-validated (Haab and McConnell, 2002).

An example of the data enriching approach is the study by Earnhart (2001) who combines a hedonic analysis (revealed preference approach) with a choice-conjoint analysis (state preference approach), in order to increase the reliability of estimated values regarding the aesthetic benefits generated by improving the quality of coastal wetlands near residential locations. In another example Birol et al. (2006) combine a choice experiment model and a discrete-choice farm household model to produce more robust estimates of the value of Hungarian agricultural biodiversity, which comprises private use values of agrobiodiversity managed in home gardens as they accrue to the farmers who manage them.

Another complementary option is the use of a ‘preference calibration’ approach in which multiple value estimates for ecosystem services and biodiversity arising from different valuation methods such as hedonic property value, travel cost demand, and contingent valuation, can be used to calibrate a single preference function to reconcile potential differences (Smith et al., 2002). This is akin to the use of specific preference restrictions to link contingent valuation estimates of environmental quality improvements to revealed preference measures for a closely selected value change, taking place for the same biodiversity component or ecosystem service. The idea is to isolate restrictions linking the parameters estimated with the different revealed (and stated preference) methods (e.g., Smith et al., 2003).

4 Insurance value, resilience and (quasi-)option value

The insurance value of an ecosystem (see section 2.3) is dependent on and related to the system's resilience. A general measure of the resilience of any system is the conditional probability that it will flip from one stability domain to another, given the current state of the system and the current disturbance regime (Perrings, 1998). These regimes are separated by thresholds, which are given by the level of disturbance that triggers a dramatic change in the state of ecosystems and the provision of ecosystem services (Luck, 2005; Muradian, 2001). Resilience relates to the vulnerability of a system, its capacity in a given state to accommodate perturbations without losing functionality (Box 4). For this section, ecological resilience is the capacity of a system to remain in a given configuration of states – a regime – in systems where multiple regimes are possible (Walker et al., 2006).

Box 4: Biodiversity and resilience

Resilience is a complex ecosystem property that is simultaneously related to the system's inner functioning and to cross-scale interactions (Holling, 2001; Holling and Gunderson, 2002). The semantics of resilience can be confusing, but studies suggest that resilience relates to features such as functional diversity within an ecosystem (Schulze and Mooney, 1993; Folke et al., 1996), and to functional redundancy within a given ecosystem function. Changes in the set of species in an ecosystem affect its capacity to support ecosystem services under various conditions, i.e. functional redundancy. The links between biodiversity change and ecosystem functioning form a hot research topic in ecology (Loreau et al., 2003; Caldeira et al., 2005; Hooper et al., 2005; Spehn et al., 2005), as does the relationship between biodiversity and the resilience of ecological systems (Scheffer et al. 2001, 2003; Walker et al. 2004; Walker et al. 2006). Despite rising attention to these issues from ecologists, our knowledge about the functioning of regulating services and the capacity of the system to maintain functionality over a range of environmental conditions is still limited.

The literature on ecological resilience offers growing evidence of regime shifts in ecosystems when critical thresholds are reached as a consequence of either discrete disturbances or cumulative pressures (Scheffer et al., 2001; Folke, 2004; Walker and Meyers, 2004). This has been studied in a wide range of ecosystems, including among others temperate lakes (Carpenter et al., 2001), tropical lakes (Scheffer et al., 2003), coastal waters (Jansson and Jansson, 2002), and savannas (Anderies et al., 2002). When such shifts occur, the capacity of the ecosystem to underpin ecosystem services can change drastically and in a non-linear way (Folke et al., 2002).

The distance to an ecological threshold affects the economic value of ecosystem services given the sate of the ecosystem (Limburg et al., 2002). Valuation exercises cannot be carried out reliably without accounting for this distance. The reason is that when the system is *sufficiently* close to a threshold, radical uncertainty or ignorance about the potential and often non-linear consequences of a regime shift becomes a critical issue. This makes standard valuation approaches to be of little use. In other words, traditional valuation under these circumstances is unreliable at best (Pritchard et al., 2000; Limburg et al., 2002). In fact, while it may be possible to develop early warning indicators to anticipate proximity to such tipping points, available scientific knowledge has not yet progressed enough to anticipate shifts with precision (Biggs et al., 2009). This implies the existence of radical uncertainty and hence poses formidable challenges to valuation. The problem is that standard approaches to estimate the total economic value of ecosystem services is based on marginal changes over some non-critical range (Turner et al., 2003). Under such circumstances policy ought to resort to other complementary instruments such as using the safe minimum standard and the precautionary principle (Turner, 2007).

In more palatable situations where science we can still deal with uncertainty about the resilience of ecosystems, decision makers still need information about the conditions that may trigger regime shifts, the ability of human societies to adapt to these transformations, and their socio-economic implications. There are at least three questions to direct a resilience assessment of ecosystem services (Walker and Pearson, 2007):

- Can major changes in the provision of ecosystem services be triggered by the transition to alternate stable regimes in a particular ecosystem?
- If so, how will the shift to the alternate regime affect people’s valuation of ecosystem services? That is, what are the consequences, in terms of economic costs and benefits?
- What is the probability of crossing the threshold? This requires knowledge about where the threshold is, the level of current disturbance, and the properties of the system (see chapter 2).

The latter question stresses the need to adopt a dynamic approach and to take into consideration the probability of alternative states given a level of disturbance. As resilience is reduced, e.g., due to human interventions, then the probability of regime shifts (either due to natural or human-induced disturbances) will rise (Scheffer et al., 2001).

One example is the regime shift that took place in Caribbean coral reefs (from pristine coral to algae-dominated systems). A pre-shift stage, characterized by increased nutrient loading combined with intensive fishing, reduced the number of herbivorous fishes. The event that led to the regime shift was a pathogen-induced mass mortality of a species of sea urchin, *Diadema antillarum*. Had the herbivorous fish populations not been so reduced in numbers, they could have replaced the ecological function of the sea urchin in controlling the population of algae (deYoung et al., 2008). The regime shift took place during the 1980s, within a period of 1-2 years, and the new state (algae-dominated ecosystems) has lasted for more than 20 years.

There are also plenty of cases showing that invasive species, whether introduced accidentally or deliberately, can also alter ecosystems and their services drastically, sometimes leading to a total and costly ecological regime shift (Maron et al., 2006, Vitule et al., 2009, Perrings et al., 2000, Pimentel et al., 2005), whether in water (Mills et al., 1993, Knowler, 2005) or on land (Cook et al., 2007). For example, *Miconia calvescens*, introduced as an ornamental tree in the 20th Century in Hawaii, has since expanded rapidly. *Miconia* is now referred as 'the purple plague' of Hawaii, where its range covers over 1,000 km^2, including extensive mono-specific stands. It threatens watersheds, reduces biodiversity severely by driving endangered native species to local extinction and lowers recreational and aesthetic values (Kaiser, 2006)

One of the features of regime shifts in ecosystems is that the new regime may have a high level of resilience itself. Therefore the costs associated with transitioning back to the previous regime, i.e. restoration costs, may be very high. The increased probability of regime shifts that furthermore may be very hard to remediate has significant implications for the economic valuation of ecosystems. As ecosystems reach thresholds, marginal human impacts on the system will lead to increasingly uncertain non-marginal effects. Under these conditions, the reliable estimation of TEV becomes increasingly difficult - if not impossible.

4.1 What is the value of ecosystem resilience?

The value of the resilience of an ecosystem lies in its ability to maintain the provision of benefits under a given disturbance regime. The role of biodiversity in supporting an ecosystem's functions has been studied by, e.g., Perrings and Gadgil (2003) and Figge (2004). Diversity within (Haldane and Jayakar, 1963; Bascompte et al., 2002) and among species (Ives and Hughes, 2002) can contribute to a stable flow of ecosystem service benefits. Ecological systems in which there are redundant species within functional groups experience lower levels of covariance in the 'returns' on members of such groups under varying environmental conditions than do systems which contain no redundant species. A marginal change in the value of ecosystem resilience thus corresponds to the difference in the expected value of the stream of benefits that the ecosystem yields given a range of environmental conditions.

The valuation of system resilience in some state can therefore be viewed to be analogous to the valuation of a portfolio of assets in a given state (Brock and Xepapadeas, 2002). The value of the asset mix – the portfolio – depends on the covariance in the returns on the individual assets it contains. Sanchirico et al. (2008) apply financial asset management tools to multi-species fisheries, for example. They show that acknowledging covariance structures between revenues from catches of individual species can achieve a reduction in risk at no cost or loss of overall revenue.

It is worth noting that just as the value of a portfolio of financial assets depends on the risk preferences of the asset holders, so does the value of the ecosystem resilience, which depends on the risk preferences of society. The more risk averse is society, the more weight it will place on strategies that preserve or build ecosystem resilience, and the higher the value it would allocate to ecosystem configurations that are less variance prone, i.e. more resilient (Armsworth and Roughgarden, 2003).

Currently, environmental economists interested in valuing resilience of ecosystems regard it not as a property but as natural capital (stock) yielding a 'natural insurance' service (flow) which can be interpreted as a benefit amenable for inclusion in cost benefit analysis (Mäler et al., 2007, Walker et al. 2009b). An example will help illustrate how and why to value resilience as an asset.

Irrigated agriculture in many parts of the world is under threat from rising salinity. Indeed, many erstwhile productive regions are now salinized and have little value to agriculture. The cause is rising water tables which are brought about through a combination of land clearing and irrigation. The rising water table brings with it salt from deeper layers in the soil up to the surface. An example in South East Australia shows that original water tables were very deep (30 m) (Walker et al., 2009). Fluctuations in rainfall caused variations in water table depth, but these were not problematic. However, there is a critical threshold in the depth of the water table – ca. 2 m, depending on soil type. Once the water table reaches this level, the salt is drawn to the surface by capillary action. When the water table is 3 m below the surface the top meter of soil – the "stock" of top soil that determines agricultural production – is the same as when the water table was 30 m below. But it is much less resilient to water table fluctuations and the risk of salinization increases. Resilience, in this case, can be estimated as the distance from the water table to 2 m below the surface. As this distance declines, the value of the stock of productive top soil diminishes. Therefore any valuation exercise that includes only the status of the top soil stock and ignores its resilience to water table fluctuations is inadequate and misleading.

Walker et al. (2009b) have estimated a value of the resilience stock 'salinity', which reflects the expected change in future social welfare from a marginal change in resilience as given by small changes in the water table today. Resilience (*X*) is equal to the current distance of the water table to the threshold, i.e., 2 m below the surface. Let $F(X_0, t)$ be the cumulative probability distribution of a flip up to time *t* if the initial resilience is X_0 based on past water table fluctuations and environmental conditions (ie. rainfall, land clearing etc.). It is assumed that the flip is irreversible or at least very costly to reverse. Walker et al. (2009b) define $U_1(t)$ as the net present value of all ecosystem service benefits at time t if the system has not shifted at that time and $U_2(t)$ as the net present value of ecosystem service benefits in the alternate regime if the system has shifted before (or at) *t*. It can then be shown that the expected social welfare of resilience W(X_0) is

$$W(X_0) = \int_0^{\infty} [S(X_0, t)U_1(t) + F(X_0, t)U_2(t)]dt$$

The current regime is one of agriculturally productive land (non-saline) and its ecosystem service value was estimated as the net present value of all current land under production (estimated market value). The alternate regime, saline land, was assumed to yield a minimal value for the land (i.e. U_2 is a small fraction of U_1) as it will loose all agricultural productivity, which is the basis for current regional social and economic conditions. The probability that the current agricultural regime will continue, $S(X_0,t)$, was estimated from past water table fluctuations and known relationships with agricultural practices now and into the future. Estimations showed significant expected loss in welfare due to salinity.

This formulation of resilience is specific to the case study but can be generalised. It may be easily extended to deal with reversible thresholds, multiple regimes (more than two), different denominators (i.e. monetary, etc.) and more than one type of resilience. The challenge lies in determining the accurate ecological and economic data that can be used to estimate probability functions, costs, discount rates, etc which are relevant to management decisions.

4.2 Main challenges of valuing ecosystem resilience

When it comes to economic valuation, at least three issues become salient in relation to non-linear behaviour and resilience of ecosystems. First, the fact that transitions may take place in uncertain, sudden and dramatic ways imposes severe limitations on the marginalist approach that underlies most valuation methods. The majority of methods allocate economic values to changes *at the margin*, assuming that small human disturbances produce proportional changes in the condition of ecosystems and therefore in their capacity to provide ecosystem services. If thresholds effects are present, however, then an extrapolation of the economic value based on marginal changes is no longer valid. As Barbier et al. (2008) formulate it, the linearity assumption “can lead to the misrepresentation of economic values inherent in (ecosystem) services” by creating a bias to either side of the conservation-development debate.

Secondly, the capacity of ecologists both to assess the level of resilience and to detect when a system is approaching a threshold is still incipient. Contamin and Ellison (2009) point out that “prospective indicators of regime shifts exist, but when the information about processes driving the system is incomplete or when intensive management actions cannot be implemented rapidly, many years of advance warning are required to avert a regime shift”. They add that to enhance predictive capacity would normally require considerable resources and time, which usually are not available to decision makers. This is particularly the case in developing countries. In addition, what seems to be clear is that the larger the spatial scale, the higher the complexity and therefore more difficult it is to detect and predict regimes shifts (deYoung et al., 2008).

Thirdly, we often fail to learn of the benefits provided by a given species or ecosystem until it is gone (Vatn and Bromley, 1994). For example, the North American passenger pigeon was once the most populous bird species on the planet, and it population was deemed inexhaustible. However, excessive hunting led to its extinction at the beginning of the 20th century. It then became clear that passenger pigeons had been consuming untold tons of acorns. Scientists speculate that with the pigeons demise, acorns were consumed by deer and mice, leading to a boom in their populations, followed by a boom in the populations of ticks that fed on them, and finally in the populations of spirochaetes that lived in the ticks. The result was an entirely unpredictable epidemic in Lyme disease several decades after the loss of the pigeons (Blockstein, 1998).

In summary, standard valuation approaches ought to be used over the non-critical range and far from ecological thresholds. by contrast, serious constraints on traditional economic valuation methods exist when ecological thresholds are identified by science as being ‘sufficiently’ sufficiently close and when the potential irreversibility and magnitude of the non-marginal effects of regime shifts are also deemed sufficiently important, . Our ability to observe and predict the dynamics of ecosystems and biodiversity will always be limited (Harwood and Stokes, 2003) and ecosystem management strategies need to consider how we live with irreducible sources of uncertainty about future benefits. In situations of radical uncertainty resilience should be approached with the precautionary principle and safe minimum standards.

Economists have traditionally used stated preference and revealed preference techniques to determine monetary values of ecosystems (reviewed in the previous sections). When radical uncertainty is not an issue, thoughts regarding the ability of these methods to handle thresholds and resilience are still being developed and new valuation approaches that account for uncertainty have been attempted, including bioeconomic models that regard resilience as a stock and not just as a property of the ecosystem.

4.3 Dealing with (quasi-) option value

In the context of valuation of expected outcomes, the concepts of “option value” and “quasi-option value” are anchored in the expected utility theory (see section 2.3). Even if an ecosystem (or component of it) has no current use, it may have option value. Barbier et al. (2009) point out, for instance, that the future may bring human diseases or agricultural pests that are unknown today. In this case, today's biodiversity would have an option value insofar as the variety of existing plants may already contain a cure against the as yet unknown disease, or a biological control of the as yet unknown pest (Polasky and Solow, 1995; Simpson et al., 1996; Goeschl and Swanson, 2003). In this sense, the option value of biodiversity conservation corresponds to an “insurance premium” (Perrings, 1995, Baumgärtner, 2007), which one is willing to pay today in order to reduce the potential loss should an adverse event occur in the future. Accordingly, option value can be defined as “the added amount a risk averse person would pay for some amenity, over and above its current value in consumption, to maintain the option of having that amenity available for the future, given that the future availability of the amenity is uncertain” (Bulte et al., 2002: 151).

The option value assumes supply uncertainty of ecosystem services and derives from risk aversion on the part of the beneficiaries of such services. It is usually measured as the difference between the *option price*, the largest sure payment that an individual will pay for a policy before uncertainty is resolved, and the *expected consumer surplus*, which is the probability-weighted sum of consumer surpluses over all potential states of the world (Pearce and Turner, 1991). The size and sign of the option value have been subject to empirical discussions and it is found to depend on the source of uncertainty (Perman et al., 2004).[vii]

If it is possible to reduce supply uncertainty about ecosystem services by acquiring further scientific information on ecosystems over time, the notion of *quasi-option value* becomes more relevant. It is the value of preserving options for future use given expectated growth of knowledge. The quasi-option value is generally agreed to be positive if such growth of knowledge is independent of actual changes in the ecosystem (Pearce and Turner, 1991). In this case quasi-option value measures the benefit of information and remaining flexible by avoiding possibly irreversible changes.

Valuation studies that have focused on quasi-option values have largely dealt with the role of bioprospecting. This is so because the uncertainty surrounding the future commercial value of the genetic material present in ecosystems creates an incentive to conserve it (Arrow and Fisher, 1974). It is argued that as uncertainty regarding the ecosystem is resolved (i.e. as the genetic material within the system is screened) the quasi-option value of resource conservation diminishes (Barrett and Lybbert, 2000).[viii]

Bulte et al. (2002) provide a possible approach to calculating quasi-option value, in the context of non-use values of primary forest in Costa Rica. The provision of ecosystem services of the forest is uncertain but expected to be increasing, and deforestation of primary forest is thought to have an irreversible negative effect on the provision of such services. The quasi-option value of maintaining primary forests is included as a component of investment in natural capital. The uncertainty of ecosystem service supply in this case – as in many others – arises essentially from uncertain income growth rates, which affect preferences and thus demand for forest conservation, as well as from the possible future availability of substitutes for the ecosystem services supplied by the forest.

It should be clear that calculating option and quasi-option values is not straightforward. First the risk preferences of individuals need to be known. While option values are associated with degrees of risk aversion, risk neutrality is assumed to hold for quasi-option values (Bulte et al., 2002). Finding out risk preferences is not trivial, however. Additionally, experimental studies on the relation between risk preferences and economic circumstances do not support simple generalizations, particularly if individuals face extraordinarily risky environments in general (Mosley and Verschoor, 2005).

Calculating option and quasi-option values are thus perhaps one of the most problematic issues surrounding valuation of ecosystem services. However, such values may be significant especially with regard to irreversible changes to natural capital. It is important to know the extent to which ecosystem services may be demanded in the future and which ones may become unavailable. It is this information about future preferences and future availability of the services that is most highly needed to calculate option and quasi-option values.

There is increasing experimental evidence that the theory of expected utility, on which the concepts of option and quasi-option rely, is not an accurate model of economic behaviour. Analysts need to compare results of estimates produced using (modified) expected utility models with estimates are based on the prospect theory, the regret theory, and other non-expected utility models (e.g., see reviews in Rekola (2004) and Mosley and Verschoor (2005) for detailed discussions). Such alternative theories are gaining more support and previous ways to estimate (quasi-) option values may need to be revised. Individuals may choose between and value ecosystem services through alternative behavioral rules than systematically weighing probabilistic outcomes.

5 Valuation across stakeholders and applying valuation in developing countries

5.1 Valuation across stakeholders

For the economic valuation of ecosystem services, identification of relevant stakeholders is a critical issue (Hein et al. 2006). In almost all steps of the valuation procedure, stakeholder involvement is essential in order to determine main policy and management objectives, to identify the main relevant services and assess their values, and to discuss trade-offs involved in ecosystem services use or enjoyment (de Groot et al., 2006). Here, stakeholders refer to persons, organizations or groups with interest in the way a particular ecosystem service is used, enjoyed, or managed.

Stakeholder-oriented approaches in economic valuation connect valuation to possible management alternatives in order to solve social conflicts. Using stakeholder analysis in ecosystem services valuation can support the identification and evaluation of who wins and who loses when possible management strategies are implemented in a social-ecological system. Hence, identifying and characterizing stakeholders and their individual reasons for conserving different ecosystem services could help resolve conflicts and develop better policies.

Socio-cultural characterization of the stakeholders beforehand may be critical to determining these underlying factors. This characterization is, however, a largely unexplored issue in economic valuation research (Manski, 2000). As stated by Adamowicz (2004), economic valuation based on factors that influence monetary value generates more useful information than making a simple inventory of values.

Different stakeholders often attach different values to ecosystem services depending on cultural background and the impact the service has on their living conditions (Hein et al., 2006; Kremen et al., 2000). Further, goods with wider spillovers are more “public” in nature, and require contributions from a more diverse set of donors. For this reason different types of ecosystem services are valued differently as the spatial scale of the analysis varies (Hein et al., 2006; Martín-López et al., 2007). Local agents tend to attach higher values to provisioning services than national or global agents, who attach more value to regulating or cultural services

Considering spatial scales and stakeholders enhances the ability of ecosystem service valuation studies to support decision-making. The formulation of management plans that are acceptable to all stakeholders requires the balancing of their interests at different scales (Hein et al., 2006). Since different stakeholders have different interests in ecosystem services use and enjoyment (Martín-López et al., 2009b), there is a potential imbalance between the costs that arise at the local level from ecosystem management and the benefits that accrue at the national and international levels. Policy makers that are aware of these differences can implement management measures that limit or even reduce social inequities. One option that is currently widely considered is to compensate people living in or near protected areas that provide the services for their losses, through Payment for Ecosystem Services (Ferraro and Kramer, 1997). This policy instrument is presented in more detail in TEEB D1 (2009) and TEEB D2 (forthcoming).

The stakeholder approach in valuation processes entails a challenge because it requires stakeholder involvement in the entire process. It may lead to identification of knowledge gaps and research needs as the process progresses (Hermans et al., 2006). This involvement can be supported by tools of participatory analysis, as well as by deliberative monetary valuation (Spash, 2007, 2008). In using tools for participatory analysis, all stakeholder types must be fairly represented in order to prevent one stakeholder type dominating the process. Therefore, identifying and selecting organizations and stakeholders representatives is an essential part of economic valuation of ecosystem services.

Future steps of the stakeholder-oriented approach in ecosystem services valuation processes should include (1) the prioritization of stakeholders based on their degree of influence in the ecosystem services management and their degree of dependence on the ecosystem services (de Groot et al., 2006), and (2) the identification of stakeholders based on their capacity to adapt to disturbances and their governance capacity in order to identify who are able to manage in the long-term the ecosystem services provided by biodiversity (Fabricius et al. 2007).

5.2 Applying monetary valuation in developing countries

Biodiversity supports a range of goods and services that are of fundamental importance to people for health, well-being, livelihoods, and survival (Daily, 1997, MA, 2005). Often, it is the people from the poorest regions in developing economies that have the greatest immediate dependency on these stocks; such as direct reliance on natural resources for food, fuel, building material and natural medicines. Gaining a better understanding of the role of biodiversity is fundamental for securing the livelihoods and well-being of people in developing countries.

In recent years many studies have examined how people value biodiversity (Nunes and Van den Bergh, 2001; Christie et al., 2004, 2007). The majority of this work has been conducted in the developed world with only limited application in developing countries (Abaza and Rietbergen-McCracken, 1998; Georgiou et al., 2006; Van Beukering et al., 2007). In a search of the Environmental Valuation Research Inventory (EVRI) database of valuation studies (http://www.evri.ca), Christie et al. (2008) have recently identified 195 studies that aimed to value biodiversity in developing countries. This number represented approximately one-tenth of all published biodiversity valuation studies at the time. These studies were equally distributed between ‘lower middle income’ and ‘lower income’ countries, but no studies were found of valuation in the poorest ‘transition economies’. Half the studies identified were conducted in Asia, 18% in Africa and 5% in South America. It is therefore evident that there is great variability in the application of valuation in developing countries, with the poorest countries and some regions having little or no coverage.

The application of economic valuation in developing countries is clearly in its infancy. Further, it is clear that there are significant methodological, practical and policy challenges associated with applying valuation techniques in developing countries. Many of these challenges stem from the local socio-economic, political situation in developing countries which may mean that a direct transfer of methods is not appropriate. Thus, it is likely that some modification of standard approaches may be required to do good valuation studies in developing countries. The Christie et al. (2008) review of biodiversity valuation in developing countries highlights many of these challenges. Here we pay special attention to methodological, practical and policy issues.

With regard to methodological issues it should be noted that low levels of literacy, education and language creates barriers to valuing complex environmental goods, as well as creating difficulties for utilizing traditional survey techniques such as questionnaires and interviews. More deliberative and participatory approaches to data collection may overcome these issues (Bourque and Fielder, 1995, Jackson and Ingles, 1998; Asia Forest Network, 2002, Fazey et al., 2007) (see Box 6).

Many developing countries have informal or subsistence economies, in which people may have little or no experience of dealing with money. The consequence of this is that they would find it extremely difficult to place a monetary value on a complex environmental good. Some researchers have attempted to address this issue by assessing willingness to pay in terms of other measures of wealth, e.g. number of bags of rice (Shyamsundar, and Kramer, 1996; Rowcroft et al., 2004).

The majority of valuation methods have been developed and refined by researchers from developed counties. There is evidence that the current best-practice guidelines for these methods might not be appropriate for applications in developing countries. For example, the NOAA guidelines for contingent valuation suggest taxation as the most appropriate payment vehicle. However, many people in developing countries do not pay taxes, and may not trust the government to deliver policy (McCauley and Mendes, 2006).

As far as implications for practitioners of valuation studies, it should be pointed out that many developing countries are affected by extreme environmental conditions which may affect the researcher's ability to access areas or effectively undertake research (Bush et al., 2004; Fazey et al., 2007). In many developing countries there may be a lack of local research capacity to design, administer and analyze research projects. However, the involvement of local people is considered essential within the research process to ensure that local nuances and values are accounted for (Whittington, 1998; Alberini and Cooper, 2000; Bourque and Fielder, 1995).[ix]

Lastly, some of the main aspects to be kept in mind when using valuation in developing countries are about the lack of local research capacity as this may result in a lack of awareness of valuation methods. A capacity building program on these issues is considered important if developing countries are to effectively address biodiversity issues. Much of the existing biodiversity valuation research is extractive, with little input from or influence on local policy (Barton et al., 1997). Incorporating ideas from action research into valuation is seen as being essential if this type of research is to meaningfully influence policy (Wadsworth, 1998).

Box 6: Participatory valuation methods

Participatory valuation methods differ from economic valuation methods in several aspects, including the following:

- **Focus**: Participatory valuation methods ought to have a focused perspective that limits data to the needs of valuation. Collecting contextual data can be important to understand local situations but collecting extraneous or unnecessary information can waste time and confuse the purpose of the valuation objective.
- **Flexibility**: It is important to allow for the ability to adapt to changing local conditions, unanticipated setbacks during the valuation study design, and the process of developing and applying specific valuation techniques in conjunction with participants.
- **Overlapping techniques**: Participatory valuation methods gain in effectiveness when different techniques collect at least some of the same data from different participants as this makes it possible to cross-check valuation results.
- **Cooperation**: In designing and implementing valuation studies, gaining the full support of local stakeholders is important to obtain reliable information and to develop a sense of learning between all participants.
- **Sharing**: The outcome of the valuation studies needs to be communicated back to stakeholders in order to strengthen the focus of the valuation approach.

Source: Jarvis et al. (2000)

It is clear that the way people in developing countries think about the natural environment is different to that of people in developed countries. All of the issues discussed above mean that it may be extremely difficult for people from developing countries to express their valuation of ecosystem services and biodiversity as compared to people from more developed economies which usually hold different value concepts that are more closely related to market economics. Hence, standard approaches to valuation in developing countries should be taken with due caution. These issues further suggest that valuation may be more effective if (i) local researchers are used throughout the research process, and (ii) deliberative, participative and action research approaches are incorporated into the valuation methods.

6 Benefit transfer and scaling up values

6.1 Benefit transfer as a method to value ecosystem services

To estimate the value of ecosystem services one would ideally commission detailed ecological and economic studies of each ecosystem of interest. Undertaking new ecological and economic studies, however, is expensive and time consuming, making it impractical in many policy settings. Benefit (or value) transfer (BT henceforth) is an approach to overcome the lack of system specific information in a relatively inexpensive and timely manner. BT is the procedure of estimating the value of an ecosystem service by transferring an existing valuation estimate from a similar ecosystem. The ecosystem to which values are transferred is termed the “policy site” and the ecosystem from which the value estimate is borrowed is termed the “study site”. If care is taken to closely match policy and study sites or to adjust values to reflect important differences between sites, BT can be a useful approach to estimate the value of ecosystem services (Smith et al., 2002).[x]

BT methods can be divided into four categories: i) unit BT, ii) adjusted unit BT, iii) value function transfer, and iv) meta-analytic function transfer.

Unit BT involves estimating the value of an ecosystem service at a policy site by multiplying a mean unit value estimated at a study site by the quantity of that ecosystem service at the policy site. Unit values are generally either expressed as values per household or as values per unit of area. In the former case, aggregation of values is over the relevant population that hold values for the ecosystem in question. In the latter case, aggregation of values is over the area of the ecosystem.

Adjusted unit transfer involves making simple adjustments to the transferred unit values to reflect differences in site characteristics. The most common adjustments are for differences in income between study and policy sites and for differences in price levels over time or between sites.

Value or demand function transfer methods use functions estimated through valuation applications (travel cost, hedonic pricing, contingent valuation, or choice modelling) for a study site together with information on parameter values for the policy site to transfer values. Parameter values of the policy site are plugged into the value function to calculate a transferred value that better reflects the characteristics of the policy site.

Lastly, meta-analytic function transfer uses a value function estimated from multiple study results together with information on parameter values for the policy site to estimate values. The value function therefore does not come from a single study but from a collection of studies. This allows the value function to include greater variation in both site characteristics (e.g. socio-economic and physical attributes) and study characteristics (e.g. valuation method) that cannot be generated from a single primary valuation study. Rosenberger and Phipps (2007) identify the important assumptions underlying the use of meta-analytic value functions for BT: First, there exists an underlying meta-valuation function that relates estimated values of a resource to site and study characteristics. Primary valuation studies provide point estimates on this underlying function that can subsequently be used in meta-analysis to estimate it; second, differences between sites can be captured through a price vector; thirdly, values are stable over time, or vary in a systematic way; and lastly, the sampled primary valuation studies provide “correct” estimates of value.

The complexity of applying these BT methods increases in the order in which they have been presented. Unit BT is relatively simple to apply but may ignore important differences between study and policy sites. Meta-analytic function transfer on the other hand has the potential to control for differences between study and policy sites but can be complex and time consuming if an existing meta-analytic value function is not available (i.e. primary studies need to be collected, coded in a database, and a value function estimated). The complexity of the BT method does not necessarily imply lower transfer errors. In cases where a high quality primary valuation study is available for a study site with very similar characteristics to the policy site, simple unit BT may result in the most precise value estimate.

BT methods generally transfer values either in terms of value per beneficiary (e.g. value per person or household) or value per unit of area of ecosystem (e.g. value per hectare). The former approach explicitly recognises that it is people that hold values for ecosystem services whereas the latter approach emphasises the spatial extent of ecosystems in the provision of services. In practical terms it is often difficult to identify the beneficiaries of ecosystem services and many valuation methods do not produce value estimates in per person/household terms (e.g. production function approach, net factor income method). It is therefore often more practical to define values for transfer in terms of units of area.

6.2 Challenges in benefit transfer for ecosystem services at individual ecosystem sites

6.2.1 Transfer errors

The application of any of the BT methods described above may result in significant transfer errors, i.e., transferred values may differ significantly from the actual value of the ecosystem under consideration. There are three general sources of error in the values estimated using value transfer:

1. Errors associated with estimating the original measures of value at the study site(s). Measurement error in primary valuation estimates may result from weak methodologies, unreliable data, analyst errors, and the whole gamut of biases and inaccuracies associated with valuation methods.
2. Errors arising from the transfer of study site values to the policy site. So-called generalisation error occurs when values for study sites are transferred to policy sites that are different without fully accounting for those differences. Such differences may be in terms of population characteristics (income, culture, demographics, education etc.) or environmental/physical characteristics (quantity and/or quality of the good or service, availability of substitutes, accessibility etc.). This source of error is inversely related to the correspondence of characteristics of the study and policy sites. There may also be a temporal source of generalisation error in that preferences and values for ecosystem services may not remain constant over time. Using BT to estimate values for ecosystem services under future policy scenarios may therefore entail a degree

of uncertainty regarding whether future generations hold the same preferences as current or past generations.

3. Publication selection bias may result in an unrepresentative stock of knowledge on ecosystem values. Publication selection bias arises when the publication process through which valuation results are disseminated results in an available stock of knowledge that is skewed to certain types of results and that does not meet the information needs of value transfer practitioners. In the economics literature there is generally an editorial preference to publish statistically significant results and novel valuation applications rather than replications, which may result in publication bias.

Given the potential errors in applying BT, it is useful to examine the scale of these errors in order to inform decisions related to the use of value transfer. In making decisions based on transferred values or in choosing between commissioning a BT application or a primary valuation study, policy makers need to know the potential errors involved. In response to this need there is now a sizeable literature that tests the accuracy of BT. Rosenberger and Stanley (2006) and Eshet et al. (2007) provide useful overviews of this literature. Evidence from recent studies that examine the relative performance of alternative BT methods for international benefit transfers suggests that value function and meta-analytic function transfers result in lower mean transfer errors (e.g. Rosenberger and Phipps, 2007; Lindhjem and Navrud, 2007).

It is not possible to prescribe a specific acceptable level of transfer error for policy decision-making. What can be considered an acceptable level of transfer error is dependent on the context in which the value estimate is used. For use in determining compensation for environmental damage, there is likely to be a need for precise estimate of value. On the other hand, for regional assessments of the value of ecosystem services, higher transfer errors may be acceptable, particularly in cases where site specific errors cancel out when aggregated.

6.2.2 Aggregation of transferred values

Aggregation refers to multiplying the unit value of an ecosystem service by the quantity demanded/supplied to estimate the total value of that service. The units in which values are transferred (either per beneficiary or per unit area) have important implications for the aggregation of values to estimate total value. In the case that values are expressed per beneficiary, aggregation implies the estimation of the total WTP of a population by applying the individual WTP value from a representative sample to the relevant population that hold values for the ecosystem service in question. In order to do this, the analyst needs to assess what the size of the market is for the ecosystem service, i.e. identify the population that hold values for the ecosystem. In the case that values are expressed per unit of area, values are aggregated over the total area of the ecosystem in question. This approach focuses more on the supply of ecosystem services than on the level of demand and care needs to be taken that it is received services and not potential supply that is assessed.

In this case, the effect of the market size for an ecosystem service needs to be reflected in the estimated per unit area value.

Aggregation can also refer to summing up the value of different ecosystem services of the same good. Summing across all services provided by a specific ecosystem provides an estimate of the total economic value of that ecosystem. This procedure should be conducted with caution to avoid double counting of ecosystem service values. As long as the ecosystem services are entirely independent, adding up the values is possible. However, ecosystem services can be mutually exclusive, interacting or integral (Turner et al., 2004). The interaction of ecosystem services and values can also be dependent on their relative geographical position, for instance with substitutes that are spatially dependent.

Aggregation of ecosystem service values over a large number of services can result in improbably large numbers (Brown and Shogren, 1998). If the estimated value of maintaining a single ecosystem service is relatively large (say one tenth of one percent of household wealth) then summing over all ecosystem services that a household might be called upon to support might give implausibly large estimates.

6.2.3 Challenges related to spatial scale

Spatial scale is recognised as an important issue to the transfer of ecosystem service values (Hein et al., 2006). The spatial scales at which ecosystem services are supplied and demanded contribute to the complexity of transferring values between sites. On the supply-side, ecosystems themselves vary in spatial scale (e.g. small individual patches, large continuous areas, regional networks) and provide services at varying spatial scales. The services that ecosystems provide can be both on- or off-site. For example, a forest might provide recreational opportunities (on-site), downstream flood prevention (local off-site), and climate regulation (global off-site). On the demand-side, beneficiaries of ecosystem services also vary in terms of their location relative to the ecosystem service(s) in question. While many ecosystem services may be appropriated locally, there are also manifold services that are received by beneficiaries at a wider geographical scale.

Spatial scale raises a number of challenges in conducting accurate BT. Most of these challenges are dealt with in separate sub-sections but are mentioned here to highlight the cross-cutting importance of spatial scale. Consideration of the spatial scale of the provision of ecosystem services and location of beneficiaries is important for the aggregation of values to calculate the total economic value of these services and for dealing with heterogeneity in site and context characteristics. The availability and proximity of substitute and complementary ecosystem sites and services in particular has a clear spatial dimension. Spatial scale is also highly relevant to the issue of distance decay and spatial discounting.

Important spatial variables and relationships for BT can be usefully defined and modelled using GIS. Socio-economic characteristics of beneficiaries (e.g. income, culture, and preferences) that are not spatial variables *per se* can also often be usefully defined in a spatial manner (e.g. by administrative area, region or country) using GIS. There are a growing number of studies that utilise GIS in conducting BT (e.g. Lovett et al. 1997; Bateman et al., 2003; Brander et al., 2008).

6.2.4 Variation in values with ecosystem characteristics and context

Values for ecosystem services are likely to vary with the characteristics of the ecosystem site (area, integrity, and type of ecosystem), beneficiaries (distance to site, number of beneficiaries, income, preferences, culture), and context (availability of substitute and complementary sites and services). It is therefore important to recognise this variation in values and make appropriate adjustments when transferring values between study sites and policy sites with different characteristics and contexts.

The characteristics of an ecosystem will influence the value of the services it provides. For example, the extent to which vegetation in coastal marshes attenuates waves and provides protection to coastal communities from storm surges, depends upon the height of the vegetation in the water column (which varies by time of year and tide), width of the vegetation zone, density of vegetation, height of waves (which varies by storm intensity), coastal bathymetry, and other factors (Das and Vincent, 2009; Koch et al., 2009). BT methods therefore need to account for differences in site characteristics. In the case of the unit transfer method, study sites and policy sites need to be carefully matched. In the case of value function transfer and meta-analytic function transfer, parameters need to be included in the functions to control for important site characteristics. Ecosystem size is an important site characteristic and the issue of non-constant marginal values over the size of an ecosystem is discussed in this chapter.

Ecosystems often have multiple and heterogeneous groups of beneficiaries (differing in terms of spatial location and socio-economic characteristics). For example, the provision of recreational opportunities and aesthetic enjoyment by an ecosystem will generally only benefit people in the immediate vicinity, whereas the existence of a high level of biodiversity may be valued by people at a much larger spatial scale. Differences in the size and characteristics of groups of beneficiaries per ecosystem service need to be taken into account in transferring and aggregating values for each service. In conducting BT it is important to control for differences in the characteristics of beneficiaries between the study and policy sites. Again this can be done by either using closely similar sites in unit transfer or by including parameters in value functions that can be used to adjust transferred values. For example, transferred values can be adjusted to reflect differences in income by using estimated elasticities of WTP with respect to income (see for example Brander et al., 2006; Schläpfer, 2006; Brander et al., 2007; Jacobsen and Hanley, 2008).

BT should also account for important differences in context, such as differences in the availability of substitute and complementary sites and services. The availability of substitute (complementary) sites within the vicinity of an ecosystem is expected to reduce (increase) the value of ecosystem services from that ecosystem. For example, in a meta-analysis of wetland valuation studies Ghermandi et al. (2008) find a significant negative relationship between the value of wetland ecosystem services and the abundance of wetlands (measured as the area of wetland within a 50 km radius of each valued wetland site). This issue is of importance to the scaling-up of ecosystem service values.

6.2.5 Non-constant marginal values

Many ecosystem service values have non-constant returns to scale. Some ecosystem service values exhibit diminishing returns to scale, i.e. adding an additional unit of area to a large ecosystem increases the total value of ecosystem services less than an additional unit of area to a smaller ecosystem (Brander et al. 2006, 2007). Diminishing returns may occur either because of underlying ecological relationships (e.g., species-area curves) or because of declining marginal utility by users of services. In contrast, other ecosystem services such as habitat provision may exhibit increasing returns to scale over some range. For example, if the dominant goal is to maintain a viable population of some large predator, habitats too small to do so may have limited value until they reach a size large enough to be capable of supporting a viable population. It is therefore important to account for the size of the ecosystem being valued and the size of the change in this ecosystem, by for example, using estimated value elasticities with respect to size (see for example, Brander et al., 2007). The appropriateness of this approach is limited by complexities in ecosystem service provision related to non-linearities, step changes, and thresholds (see chapter 2). Simple linear adjustments for changes in ecosystem size will not capture these effects.

6.2.6 Distance decay and spatial discounting

The value of many ecosystem services is expected to decline as the distance between beneficiary and ecosystem increases (so called distance decay). The rate at which the value of an ecosystem service declines with distance can be represented by spatial discounting, i.e. placing a lower weight on the value of ecosystem services that are further away (or conversely, making a downward adjustment to estimated values held by beneficiaries that are located further from the ecosystem site).

Aggregation of transferred values across beneficiaries without accounting for distance decay may result in serious over-estimation of total values. An illustrative example can be found in Bateman et al. (2006), who compare different aggregation methods and assess the effect of neglecting distance-effects. Instead of simply aggregating sample means, they apply a spatially sensitive valuation function that takes into account the distance to the site and the socio-economic characteristics of the population in the calculation of values. Thereby, the variability of values across the entire economic market area is better represented in the total WTP. They found that not accounting for distance in the aggregation procedure can lead to overestimations of total benefits of up to 600%.

The rate of distance decay is likely to vary across ecosystem services. Direct use values are generally expected to decline with distance to an ecosystem but the rate of decay will vary across ecosystem services depending on how far beneficiaries are willing to travel to access each specific service, the differentiated availability of substitute services, or the spatial scale at which ecosystem services are ‘delivered’ by an ecosystem. The market size or economic constituency for ecosystem services from a specific ecosystem will therefore vary across services. For example, beneficiaries may be willing to travel a large distance to view unique fauna (distance decay of value is low and people in a wide geographic area hold values for the ecosystem and species of interest) whereas beneficiaries may not travel far to access clean water for swimming (distance decay of value is high due to availability of substitute sites for swimming and only people within a short distance of the ecosystem hold values for maintaining water quality to allow swimming). Non-use values may also decline with distance between the ecosystem and beneficiary, although this relationship may be less related to distance than to cultural or political boundaries. The spatial discounting literature suggests that non-use values should have much lower spatial discount rates than use values (Brown et al., 2002). In some cases, non-use values may not decline at all with distance, i.e. the rate of spatial discounting is zero. This might be the case for existence values for certain charismatic species that are known worldwide.

Loomis (2000) examines spatial discounting for the preservation of a range of threatened environmental goods in the US (spotted owls, salmon, wetlands, as well as a group of 62 threatened and endangered species). The first finding from this research is that WTP does fall off with distance. However, there are still substantial benefits to households that live more than a thousand miles from the habitat areas for these species. This implies that limiting summation of household benefits to nearby locations results in a large under-estimation of the total benefits. These results have two implications for BT. First, WTP is not zero as one moves beyond commonly used political jurisdictions such as states in the U.S. and possibly within single countries in the European Union. Given the available data there are no means to ascertain how values change across countries. Such cross-country comparison of values of ecosystem services is an important avenue for future research. Second, while values per household do not fall to zero at distances of a thousand miles or more, it is important to recognize that there is a spatial discount, so generalizing values obtained from an area where the species resides to the population in a wider geographic area would overstate WTP values. The limited data discussed above suggests there may be a 20% discount in the values per household at 1,000 miles and a 40% to 50% discount at 2,000 miles for high profile species or habitats.

6.2.7 Equity weighting

In conducting BT between study and policy sites with different socio-economic characteristics it is important to take account of differences in income levels. Generally there is an expectation that WTP for environmental improvements is positively related to income. Adjustments to transferred values can be made using estimated income elasticities (e.g., Brander et al., 2006; Schläpfer, 2006; Brander et al., 2007; Jacobsen and Hanley, 2008). An argument can also be made, however, for the use of equity weighting to reflect the greater dependence of the poor, particularly in developing countries, on ecosystem services, specially provisioning services (food and shelter). Equity weights correspond to the intuition that ‘a dollar to a poor person is not the same as a dollar to a rich person’. More formally, the marginal utility of consumption is declining in consumption: a rich person will obtain less utility from an extra dollar available for consumption compared to a poor person.

Equity-weighted ecosystem service value estimates take into account that the same decline in ecosystem service provision to someone who is poor causes greater welfare loss than if that change in service had happened to someone who is rich. Using local or regional data instead of national data for such an exercise is important in order to avoid smoothing of income inequalities by using larger regions to calculate average per capita incomes. Use of equity weights is particularly appropriate in the context of transferring values for ecosystem services from developed to developing countries, given the huge difference in income of those effected and the difficulties to assess the true welfare loss (Anthoff et al., 2007).

6.2.8 Availability of primary estimates for ecosystem service values

The scope for using BT for estimating the value of ecosystem services is limited by the availability of high quality primary valuation studies for all relevant ecosystem types, ecosystem services, and socio-economic and cultural contexts. Importantly, data from poorly designed empirical studies will compromise the robustness of BT (the phrase ‘*garbage in, garbage out’* appropriately describes this issue). Some types of ecosystem are well-represented in the economic valuation literature (e.g. wetlands and forests) whereas for others there are relatively few primary valuation studies from which to transfer values (e.g. marine, grassland and mountain ecosystems). Similarly, some ecosystem services are better covered in the valuation literature than others. For example, recreation and environmental amenities are well-represented whereas valuation studies for regulating services are uncommon. There is also a relative dearth of ecosystem service valuation studies conducted in developing countries (Christie et al., 2008). This represents a major gap in the available information base for BT since dependence on and preferences for ecosystem services, and consequently values, are likely to be substantially different between developed and developing countries.

There is also a (understandably) limited availability of primary valuation studies that estimate values for changes in ecosystem services outside of the context of the current availability of substitute and complementary ecosystems. The marginal value of changes in ecosystem service provision in a situation where the overall level of provision is greatly diminished is therefore beyond the domain of general observations and therefore principally unknown. This has implications for the possibilities for scaling-up ecosystem services values across large geographic areas and entire stocks of ecosystems.

6.3 Scaling-up the values of ecosystem services

The challenges encountered in conducting reliable BT discussed above relate to the transfer of values to estimate the value of individual ecosystem sites. When using BT to estimate the value of an entire stock of an ecosystem or provision of all ecosystem services within a large geographic area (so-called ‘scaling-up’), the value of ecosystem services over an entire region or biome cannot be found simply by adding up estimated values from smaller ecosystem sites, a problem that becomes much worse in the presence of nonlinear socioecological dynamics. Large-scale changes in the provision of ecosystem services will likely result in changes in the marginal value of services. Therefore, scaling-up to estimate the total economic value in a large geographic area requires taking account of the non-constancy of marginal values. Adjustments to these values can be made using estimated value elasticities with respect to ecosystem scarcity (e.g., Brander et al., 2007).

Conceptually, the economic value of a loss in the provision of an ecosystem service can be expressed as the area under the demand curve for the service that is bounded by the pre-change level of provision and the post-change level of provision, everything else being equal. For some ecosystem services it may be possible to make general assertions about the shape of the demand curve. It is possible to make general assertions about the shape of the demand curve for some ecosystem services. For example, in cases where ecosystem services can be relatively easily and cheaply provided through human-engineered solutions, or degraded or lost without much loss of utility, the demand curve should be relatively easy to draw. However, for critical services essential to sustain human life and for which no adequate substitutes are available (Ekins et al., 2003a; Farley, 2008) such estimations are much harder. Therefore, our capacity to predict future demand for scarcer environmental goods or services, whose dynamics moreover are hardly predictable, will likely remain very limited.

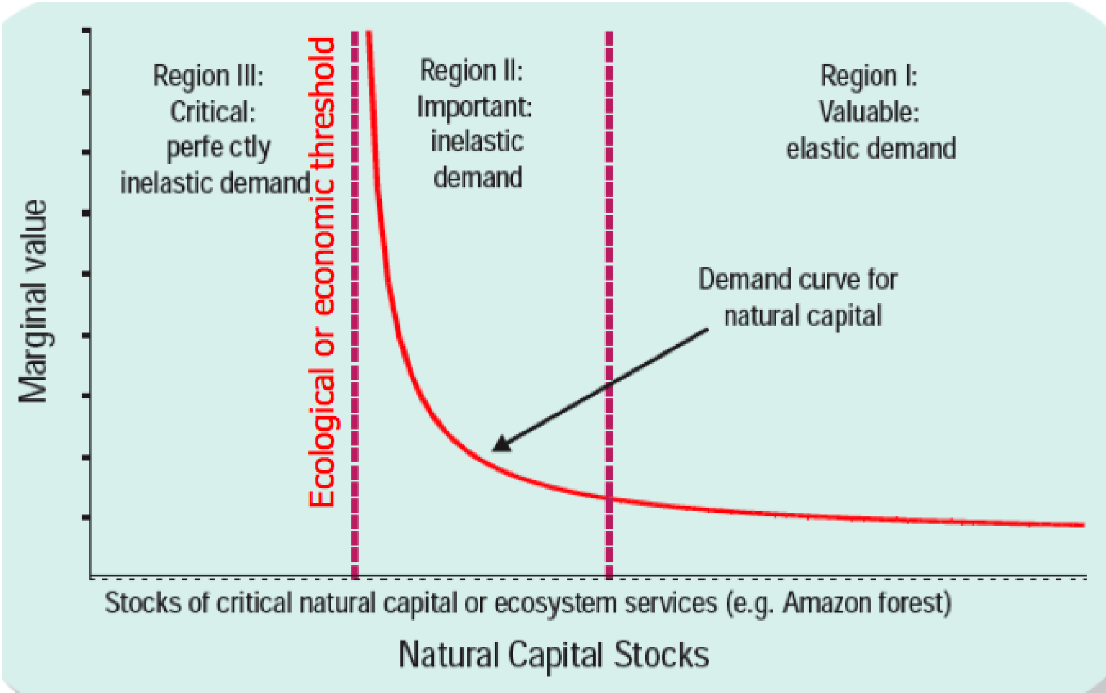


Figure 5. **The demand curve for natural capital** (Farley, 2008) Figure 5 depicts a stylized demand curve for critical natural capital with an economic or ecological threshold.

In region 1, where stocks are abundant and marginal value is low, marginal values remain reasonably constant with respect to changes in stocks. Over this range of service provision, the value of changes in supply can be reasonably well estimated using constant marginal values. Monetary valuation may facilitate decisions on allocation between conserving or not natural capital. As the overall level natural capital declines (region II), marginal value begins to rise steeply and natural capital stocks are less resilient and approaching a threshold beyond which they cannot spontaneously recover from further loss or degradation. Marginal uses are increasingly important, and values are increasingly sensitive to small changes in stocks (inelastic demand). Hence, over this range the use of *constant* marginal values to assess changes in ecosystem service supply could result in large errors in valuation (usually underestimates given that currently observed marginal values are low but risings). It is thus risky to transfer constant values from a site associated with a level of capital in region I to another site associated with region II. Further as Farley (2008) notes, conservation needs should determine the supply of the natural stock available for being exploited and hence its price. In region III, capital stocks have passed critical ecological thresholds. If not close subsitute for such ecosystem exists for those valuing it, marginal values are essentially infinite, and restoration of natural capital stocks essential (Farley, 2008). In region III, standard valuation techniques, including benefit transfer are not useful any more

The problem of dealing with non-constant marginal values over large changes in the stock of an ecosystem becomes more difficult in the presence of nonlinear ecological dynamics. Similar to the difficulty in accounting for threshold effects in valuing (and transferring values to) individual ecosystem sites, we lack knowledge of how ecosystem service values change following large-scale losses. The difficulties of conventional micro-economic methods in dealing with these complexities call for alternative approaches to be combined with TEV-based approaches which would aid decision processes at higher scales, such as deliberative and multicriteria methods (Spash and Vatn, 2006).

Current available studies measure the value of ecosystem services around present levels of overall provision (studies usually focus on one ecosystem site, with the implicit or explicit assumption that the level of provision of services from the remaining stock of ecosystems is not changed). Large changes in the overall level of provision are therefore beyond the domain of our observations and are therefore principally unknown. This makes the assessment of the value of large or complete loss of an ecosystem service impossible. Crossing ecological thresholds in critical natural capital (region III) may involve large changes welfare that render the estimation of marginal and total values essentially meaningless since they approach infinity. Scaling-up ecosystem service values across a range of service provision may be possible, particularly if adjustments are made to reflect non-constancy of marginal values over the stock, but it is important to recognise the limitations of this approach to estimate the value of large scale or complete losses of (critical) natural capital.

7 Conclusions

This chapter has addressed some of the most important theoretical and practical challenges of assessing the economic value of ecosystem services. For example, it has tackled some critical issues regarding the way values may be scaled up geographically to offer total value for ecosystem services for ecosystems, regions, biomes or indeed the entire world, an approach upon which other chapters (7 and 8) of the TEEB report are based. It has also addressed some of the most important challenges for valuation studies, especially with regard to confronting problems such as high uncertainty and ignorance and taking into consideration dynamic behavior of ecosystems.

The role of valuation and the TEV approach

This chapter has provided an overview on the rationale behind economic valuation of ecosystem services, the available methods and tools, and some key challenges. Since many ecosystem services are produced and enjoyed in the absence of market transactions, their value is often underestimated and even ignored in daily decision-making. One of the ways to tackle this information failure and make the value of ecosystems explicit in economic decision-making is to estimate the value of ecosystem services and biodiversity in monetary terms. We have suggested that the economic value of ecosystems resides basically in two aspects. The first is the total economic value of the ecosystem service benefits at a given ecological state. The second is the insurance value that lies in the resilience of the ecosystem, which provides flows of ecosystem service benefits with stability over a range of variable environmental conditions.

The value of ecosystems is generally estimated using the so-called total economic value (TEV) approach. The TEV of an ecosystem is generally divided in use- and non-use values, each of which can be further disaggregated in several value components. Valuation methods that follow the TEV approach can be divided into three main categories, direct market approaches, revealed preferences and stated preference techniques, the latter of which is being increasingly combined with deliberative methods from political science to develop formal procedures for deliberative group valuation of ecosystem values. These have been described briefly, discussing some of their strengths and weaknesses, as well as some of the aspects that have been subject to criticism.

Through the use of synthesis tables, each method has been analyzed in terms of its relative capacity to deal with specific value components and types of ecosystem services. An extensive literature data base has also been provided specifically for the key biomes forest and ecosystems. Building on a case study data base, we have reviewed how these biomes have been treated in the literature on economic valuation of ecosystems and provided quantitative data on which specific methods have been used for specific ecosystems services and value types. This chapter has also addressed several challenges valuation practitioners are faced with when adapting valuation methods to various institutional and ecological scales, such as valuation across stakeholders and applying valuation methods in developing countries.

The role of uncertainty

Regarding uncertainty inherent to valuation methods, this chapter has dealt with various types of uncertainty**.** The standard notion of uncertainty in valuation conflates risk and Knightian uncertainty. This chapter has also acknowledged the more profound type of uncertainty, here called ‘radical uncertainty’ or ‘ignorance’. This chapter has discussed ways in which the standard concept of uncertainty is applied in the valuation of ecosystem services and biodiversity and the implications of recognising radical uncertainty especially in the case of dealing with ecological resilience.

In addition, three sources of uncertainty pervading valuation of ecosystem services and biodiversity have been taken into account: (i) uncertainty regarding the delivery or supply of ecosystem services and biodiversity, (ii) preference uncertainty and (iii) technical uncertainty in the application of valuation methods.

The uncertainty regarding the delivery of ecosystem services makes stated preference methods complex. This may be the reason why there are few examples where stated preference approaches have considered the issue of uncertainty in an explicit way. Stated preference methods have generally resorted to measuring respondents’ risk perceptions. Other valuation approaches based on expected damage functions are based on risk analysis instead.

Preference uncertainty is inversely related to the level of knowledge and experience with the ecosystem service to be valued. This source of uncertainty has been more widely acknowledged in stated preference approache. for instance by requesting respondents to report a range of values rather than a specific value for the change in the provision of an ecosystem service.

Lastly, technical uncertainty pervades valuation studies specially with regard to the credibility of the estimates of non-use values through stated preference methods and the non conclusive issue of the large disparity between WTP and WTA value estimates. It has been suggested that combining valuation models and a preference calibration approach may be the way forward to minimise technical uncertainty.

The value of ecosystem resilience

The discussions in this chapter mostly address contemporary economic valuation techniques and estimates produced with these techniques. However, it should be borne in mind that these valuation techniques, which assume smooth and small system changes, may produce meaningless results in the context of ecosystems characteristics and dynamics such as ecological thresholds, resilience and regime shifts. Addressing these issues remains an important challenge in environmental valuation. Further advancements in these fields would require both a better knowledge of ecological processes and innovative valuation techniques.

The value of the resilience of an ecosystem is related to the benefits and costs that occur when the ecosystem shifts to another regime. An analogy can be drawn between the valuation of ecosystem resilience and the valuation of a portfolio of assets in that the value of the asset mix – the ecosystem and its biodiversity – depends on the probability that a shift occurs as well as the benefits and costs when it does. Current knowledge about biodiversity and ecosystem dynamics at this point is insufficient to implement such portfolio assessment and monetary analysis will be misleading when ecosystems are near critical thresholds. At the policy level, it is better to address this uncertainty and ignorance by employing a safe minimum standard approach and the precautionary principle.

Using benefit transfer

With regard to the use of secondary data, the approach of value or benefit transfer (BT) has been discussed, both in terms of its main advantages and limitations**.** BT is the procedure of estimating the value of one ecosystem (the ‘policy site’) by transferring an existing valuation estimate from a similar ecosystem (the ‘study site’). BT methods can be divided into four categories in increasing order of complexity: i) unit BT, ii) adjusted unit BT, iii) value function transfer, and iv) meta-analytic function transfer. BT using any of these methods may result in estimates that differ from actual values, so-called transfer errors. The acceptable level of transfer error for decision-making is context-specific, but if a highly precise value estimate is required it is recommended to commission a primary valuation study.

BT can be a practical, timely and low cost approach to estimate the value of ecosystem services, particularly for assessing policy scenarios involving a large number of diverse ecosystems. However, since marginal values are likely to vary with ecosystem characteristics, socio-economic characteristics of beneficiaries, and ecological context, care needs to be taken to adjust transferred values when there are important differences between study and policy sites.

Important site characteristics include the type of ecosystem, the services it provides, its integrity and size. Beneficiary characteristics include income, culture, and distance to the ecosystem. It is important to account for distance decay effects in determining the market size for an ecosystem service and in aggregating per person values across the relevant population. It should be noted that the market size and rate of distance decay is likely to vary across different ecosystem services from the same ecosystem. It is also important to account from differences in site context in terms of the availability of substitute and complementary ecosystems and services.

In cases where a high quality primary valuation study is available for a study site with very similar characteristics to the policy site, the unit transfer method may produce the most precise value estimate. In cases where no value information for a closely similar study site is available, value function or meta-analytic function transfer provide a sound approach for controlling for site specific characteristics.

Transferred values are generally expressed either per beneficiary or per unit of area. The former focuses the analysis on the demand for the service and the latter focuses on the supply. Aggregation of transferred unit values across the relevant population or ecosystem area needs to be undertaken carefully to avoid double counting values or misspecifying the market size for an ecosystem service.

Scaling-up refers to the use of BT to estimate the value of an entire stock of an ecosystem or provision of all ecosystem services within a large geographic area. In addition to the other challenges involved in using BT, scaling-up values requires accounting for the non-constancy of marginal values across the stock of an ecosystem. Simply multiplying a constant per unit value by the total quantity of ecosystem service provision is likely to underestimate total value. Appropriate adjustments to marginal values to account for large-scale changes in ecosystem service provision need to be made, for example by using estimated elasticities of value with respect to ecosystem scarcity. This approach may be useful for estimating total values over a certain range of ecosystem service provision but is limited by non-linearities and thresholds in the underlying ecological functions, particularly in the case of critical natural capital.

Final words

It should become clear that techniques to place a monetary value on biodiversity and ecosystem services are fraught with complications, only some of which currently can be addressed. Despite these limitations, demonstrating the approximate contribution of ecosystems to the economy remains urgently needed and the contribution of this chapter should be understood in this light. Valuation exercises can still provide information that is an indispensable component of environmental policy in general. As Kontoleon and Pascual (2007) state, ignoring information from preference-based valuation methods is thus neither a realistic nor a desirable option. Instead, policy-makers should interpret and utilize the valuable information provided by these techniques while acknowledging the limitations of this information.

In this context, chapters 7 and 8 of this report intend to show policy makers that there is a probability of massive losses due to depletion of natural capital. The closer we believe we are to a threshold, the more important it is to improve valuation methods to estimate what is at stake. This will emphasise the importance of ensuring that natural capital stocks remain far from critical thresholds. It is likely that new techniques and combinations of different methodological approaches (e.g., monetary, deliberative and multicriteria methods) will be needed in order to properly face future challenges and provide more accurate values that would benefit decision-making processes.

Koch et al. (2009) call for such a new decision-making approach to ecosystem services management. They recommend a number of actions that have to be taken to move in that direction, among them filling existing data gaps, especially using comparative studies; to develop ecological modelling to understand patterns of non-linearity across different spatial and temporal scales; and to test the validity of assumptions about linearity in the valuation of ecosystem services at different scales. A closer collaboration between ecologists and economists may then contribute to develop valuation techniques that are better suited to dealing with the complex relationship between ecosystems and the services they provide to the local and global economies. Last but not least, future valuation practitioners of biodiversity and ecosystem services should make explicit the procedures and methods used in their studies as well as openly acknowledge any obstacles that they may have encountered.

[ii] Economists usually conflate risk and uncertainty (in the Knightian sense). For instance Freeman (1993: 220) defines 'individual uncertainty' to "situations in which an individual is uncertain as to which of two or more alternative states of nature will be realized". In this chapter the terms risk and uncertainty are used in a conflated way following Freeman (1993) but the different type of 'radical uncertainty' or 'ignorance' due to science is also acknowledged explicitly.

[iii] A number of studies have used information on uncertainty with regard to preferences to shed light about the disparity between hypothetical values and actual economic behaviour (e.g., Akter et al., 2008).

[iv] See Akter et al. (2008) for a theoretical framework based on cognitive psychology to select explanatory variables in econometric models aimed at explaining variations in preference uncertainty beyond the more intuitive variables.

[v] An alternative strand assumes that there is an "underlying vagueness of preferences" and uses fuzzy theory to address both lack of accurate understanding of what is the nature of the ecosystem service and uncertainty about the values that have already been measured (Van Kooten et al., 2001: 487).

[vi] The National Oceanic and Atmospheric Administration (1993), or better known as the "NOAA" panel was chaired by Nobel laureates in economics such as Kenneth Arrow and Robert Solow.

[vii] If only the supply of the good is uncertain, the option value is positive if assumed that individuals are risk averse (Pearce and Turner, 1991). If other sources of uncertainty also exist, such as preference uncertainty, the sign of the option value is indeterminate.

[viii] Most studies that have focused on the value of bioprospecting are based on benefit-cost analysis by allowing explicit weights to various opportunity cost, such as land conservation, as opposed to the option value or expected benefits from the 'discovery' of a useful property of a given genetic material, net of the associated research and development costs such as biological material screenings (Pearce and Purushothaman, 1992; Simpson et al., 1996; Rausser and Small, 2000; Craft and Simpson, 2001).

[ix] There is some evidence that it may be easier to do valuation studies in developing countries (Whittington, 1998): response rates are typically higher, respondents are receptive to listening and consider the questions posed, and interviewers are relatively inexpensive (allowing larger sample sizes).

[x] An alternative approach to BT is based on "preference calibration" but this is a much more information intensive approach and thus this chapter does not cover it (see: Smith et al. 2002).

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ANNEX A

Applied sources for technical support on biodiversity valuation for national agency teams

There are two types of readily available sources of technical support on biodiversity valuation for national policy teams:

a) Applied literature on targeted valuation methods. Indicative non technical reference manuals on valuation techniques such as:

Dixon, John Louise Scura, Richard Carpenter and Paul Sherman (1994) *Economic Analysis of Environmental Impacts*, Earthscan

Bateman, I., *et al*. (2002), *Economic Valuation With Stated Preference Techniques: A Manual*, Edward Elgar.

As well as useful technical support web-sites such as

www.biodiversityeconomics.org

http://www.ecosystemvaluation.org/default.htm

http://envirovaluation.org/

b) Data-bases of existing valuation studies and data including:

EVRI - Environmental Valuation Reference Inventory: http://www.evri.ca/

ENVALUE environmental valuation database: http://www.epa.nsw.gov.au/envalue/

Valuation Study Database for Environmental Change: http://www.beijer.kva.se/valuebase.htm
The New Zealand Non-Market Valuation DataBase
http://learn.lincoln.ac.nz/markval/

RED Data Base: http://www.red-externalities.net/

Benefit transfer information pages
http://www.idrc.ca/en/ev-73300-201-1-DO_TOPIC.html
http://yosemite.epa.gov/EE/epa/eed.nsf/webpages/btworkshop.html

ANNEX B

Table A1.a Conceptual matrix based on wetland ecosystem services, benefits/value types and valuation approaches:

WETLAND SERVICES	*Stated Preference*	*Revealed Preference*	*Production based*	*Cost based*	*Benefits Transfer*
PROVISIONING					
Food *(e.g. Production of fish, wild game/hunting, fruits and grains)*	***Choice modelling*** *Layton et al. 1998; Seferlis 2004; Psychoudakis et al. 2004; Carlsson et al. 2003)* ***Contingent ranking*** *(e.g. Emerton1996)* ***CVM*** *(e.g. Bergstrom, 1990; Hammack and Brown, 1974; Benessaiah 1998; Hanley and Craig 1991)* ***Participatory Valuation*** *(e.g. L.Emerton, 2005; IUCN-WANI, 2005;)* ***Stakeholder Analysis and CVM*** *(e.g. Bhatta, 2000)*		***Factor Income/Production Function*** *(e.g., Barbier, Adams and Kimmage 1991; Barbier et. al., 1993; Hammack and Brown, 1974; Costanza et al., 1997; Hodgson and Dixon,1988; Emerton, 1998; Bann 1999; Gammage 1997; Barbier and Strand 1998; Janssen and Padilla 1997; Nickerson 1999; Verma et al. 2003; Khalil, 1999; Emerton 2005; Stuip et al. 2002; Benessaiah, N. 1998; Ruitenbeek, 1994;)*	***Opportunity cost*** *(e.g. Dixon and Sherman 1990; Hodgson and Dixon, 1988; Kramer et al.1992, 1995; Emerton, 2005, Ruitenbeek 1989a, 1989b;)* ***Public Investments*** *(e.g Powicki 1998 ; Emerton, 2005)* ***Replacement cost*** *Gren et al. 1994; Abila 1998)* ***Restoration cost*** *(e.g. Verma et al. 2003)*	***Benefits Transfer*** *(e.g.White et al. 2000; Stuip et al. 2002; Costanza et al., 1997)*
Water *(e.g. Storage and retention of water for domestic, industrial and agricultural use)*	***Choice modelling*** *(e.g. Gordon et al. 2001;)* ***CVM for non-user benefits*** *(e.g. James and Murty 1999;)* ***Participatory Valuation*** *(e.g. IUCN-WANI 2005)*	***Public Investments*** *(e.g. Powicki 1998; Emerton, 2005)*	***Factor Income*** *(e.g. Emerton, 2005)*	***Opportunity cost*** *(e.g. Emerton, 2005)* ***Replacement cost*** *(e.g. Gren et al. 1994)* ***Restoration cost*** *(e.g. Verma et al. 2003; Emerton 2005)*	
Raw Materials*(e.g. fibres, timber, fuelwood, fodder, peat,fertiliser, construction material etc.)*	***Contingent ranking*** *(e.g. Emerton 1996)* ***CVM*** *(e.g. Hanley and Craig 1991)* ***Participatory Valuation*** *(e.g. Eaton, 1997; Emerton 2005; IUCN-WANI 2005)*	***Public Investments*** *(e.g Powicki 1998; Emerton, 2005)*	***Factor Income*** *(e.g. Khalil,1999;Ruitenbeek, 1994; Verma et al. 2003; Emerton, 2005; Stuip et al. 2002,)*	***Opportunity cost*** *(e.g. Emerton, 2005)* ***Replacement cost*** *(e.g. Gren et al. 1994,)* ***Restoration cost*** *(e.g. Emerton, 2005; Verma et al. 2003)*	
Genetic resources *(e.g. biochemichal production*	***Participatory Valuation*** *(e.g. Emerton 2005; IUCN-WANI*		***Bioeconomic Modelling*** *(e.g. Hammack and Brown, 1974)*		

WETLAND SERVICES	***Stated Preference***	***Revealed Preference***	***Production based***	***Cost based***	***Benefits Transfer***
models and test-organisms, genes for resistance to plant pathogens)	*2005)*				
Medicinal resources *(e.g extraction of medicines and other materials from biota)*	***Participatory Valuation*** *(e.g. L.Emerton, 2005; IUCN-WANI, 2005;)*			***Avoided cost*** *(e.g. Emerton, 2005)* ***Restoration cost*** *(e.g. Emerton, 2005)*	
Ornamental resources species *(e.g aquarium fish and plants like lotus)*	***Participatory Valuation*** *(e.g. Emerton, 2005)*		***Factor Income*** *(e.g. Vidanage et al. 2005)*		
Human Habitat *(e.g forest provide houding to many dwellers)*				***Conversion Cost*** *(e.g. Abila, 1998)*	
Transport *(e.g Wetlands are source of navigation)*					
REGULATING					
Air quality regulation *(e.g., capturing dust particles*					
Climate regulation *(e.g. Source of and sink for greenhouse gases; influ-ence local and regional temperature, precipitation, and other climatic processes incl. carbon sequestration)*	***Participatory Valuation*** *(e.g. Emerton, 2005)*			***Avoided cost*** *(e.g. Emerton, 1998; Emerton, 2003;)*	
Moderation of extreme events *(e.g. storm protection, flood prevention, coastal protection, fire prevention)*	***CVM*** *(e.g. Hanley and Craig 1991; Bateman et al. 1993)* ***Participatory Valuation*** *(e.g. Emerton, 2005)*			***Avoided cost*** *(e.g. Bann 1999; Costanza et.al. 1997)* ***Replacement Cost*** *(e.g. Gupta 1975; Farber 1987)*	
Regulation of water flows/ Hydrological regimes *(natural drainage, flood-plain function, storage of water for agriculture or industry, drought prevention*	***Choice modelling*** *Adamowicz et al. 1994; Birol et al. 2007; Ragkos et al. 2006;)* ***Participatory Valuation*** *(e.g. Emerton, 2005; IUCN-WANI, 2005;)*		***Factor Income*** *(e.g. Acharya, 2000;)*	***Avoided cost*** *(e.g. L.Emerton, 2005)* ***Replacement cost*** *(e.g. Grenet al. 1994)* ***Restoration cost*** *(e.g. Emerton, 2005)*	

WETLAND SERVICES	***Stated Preference***	***Revealed Preference***	***Production based***	***Cost based***	***Benefits Transfer***
groundwater recharge/discharge)					
Water purification/detoxification , and waste treatment/pollution control *(e.g. retention, recovery, and removal of excess nutrients and other pollutants)*	***CVM*** *(e.g. Gren, 1995)*		***Factor Income*** *(e.g. Gren, 1995)*	***Avoided costs*** *(e.g Verma et al. 2003)* ***Mitigation Cost*** *(e.g. Sankar (2000)* ***Replacement cost*** *(e.g. Emerton 2005; Gren et al. 1994; IUCN 2003; Stuip et al. 2002;)* ***Restoration cost*** *(e.g. Gren 1995; Verma et al. 2003)*	
Erosion prevention *(e.g. retention of soils and sediments)*	***CVM*** *(e.g. Hanley and Craig, 1991;Bateman e.al, 1993; Loomis, 2000;)* ***Participatory Valuation*** *(e.g. Emerton, 2005)*				
Soil formation /conservation *(e.g. sediment reten-tion and accumula-tion of organic matter)* ***Note:*** *should come under support services*	***Choice modelling*** *Colombo et al. 2004; Colombo et al. 2006;)* ***CVM*** *(e.g. Loomis, 2000)*			***Restoration cost*** *(e.g. Emerton, 2005)*	
Pollination *(e.g. habitat for pollinators)*			***Factor Income*** *(e.g. Seidl, 2000)*		
Biological control *(e.g. seed dispersal, pest species and disease control)*					
HABITAT/SUPPORT					
Biodiversity and Nursery service *(e.g. habitats for resident or transient species)*	***Choice modeling*** *(e.g. Brouwer et al. 2003)*			***Replacement cost*** *(e.g. Gren et al. 1994)*	
Gene pool protection/ endangered species	***CVM*** *(e.g. Eija Moisseinen 1993)*			***Replacement cost*** *(e.g. Gren et al. 1994)*	

WETLAND SERVICES	***Stated Preference***	***Revealed Preference***	***Production based***	***Cost based***	***Benefits Transfer***
proetction					
Nutrient cycling *(e.g. Storage, recycling, processing, and acquisition of nutrients)*				***Replacement cost*** *(e.g. Gren et al. 1994)*	***Benefits Transfer*** *(e.g. Andréassen-Gren & Groth 1995)*
CULTURAL					
Aesthetic *(e.g. appreciation of natural scenery, other than through deliberate recreational activities)*	***Choice modelling*** *(e.g. Bergland 1997)* ***CVM*** *(e.g. Mahan 1997)*	***Hedonic pricing*** *(e.g. Verma et al. 2003; Mahan 1997)*	***Replacement Cost*** *(e.g. Gupta, 1975)*		
Recreation & tourism/ Ecotourism, Wilderness (remote-non-use) *(e.g. Opportunities for tourism and recreational activities)*	***Choice modelling*** *(e.g. Boxall et al. 1996; Carlsson et al. 2003; Hanley et al. 2002; Horne et al. 2005; Boxall and Adamomicz 2002; Adamowicz et al. 1994;* *Adamowicz et al. 1998b)* ***CVM*** *(e.g. Thibodeu & Ostro 1981; Naylor & Drew 1998; Murthy & Menkhuas, 1994; Manoharan 1996; Costanza et al. 1997; Manoharan and Dutt, 1999; Maharana et al. 2000; Wilson & Carpenter 2000; Stuip et al. 2002; Bergstrom 1990; Bell 1996; Pak and Turker, 2006)* ***Participatory Valuation*** *(e.g. IUCN-WANI 2005)*	***Consumer Surplus*** *(e.g. Bergstrom et al. 1990)* ***TCM*** *(e.g. Farber 1987; Chopra 1998; Hadker et al. 1995; Manoharan 1996; Pak and Turker, 2006; Willis et al. 1991)*		***Opportunity Cost*** *(e.g. Loomis et al. 1989)* ***Protection cost*** *(e.g. Pendleton 1995)* ***Replacement and Conversion Cost*** *(e.g. Abila 1998)*	***Benefits transfer*** *(e.g. Sorg and Loomis 1984; Walsh et al. 1988; MacNair 1993; Loomis et al. 1999; Markowski et al. 1997; Rosenberger and Loomis 2000; Andréassen-Gren & Groth, 1995)*
Educational *(e.g. Opportunities for formal and informal education and training)*					
Spiritual & artistic inspiration *(e.g. source of inspi-ration; many reli-gions attach*	***CVM*** *(e.g. Maharana et. al., 2000)*				

WETLAND SERVICES	***Stated Preference***	***Revealed Preference***	***Production based***	***Cost based***	***Benefits Transfer***
spiritual, scared and religious values to aspects of wetland and forest ecosystems)					
Cultural heritage and identity *(e.g. sense of place and belonging)*	***Choice modelling*** *(e.g. Tuan et al. 2007)* ***CVM*** *(e.g. Shultz et al. 1998; Tuan et al. 2007)*				
Information for cognitive development					

Table A1.b Conceptual matrix based on forest ecosystem services, benefits/value types and valuation approaches

FOREST SERVICES	*Stated Preference*	*Revealed Preference*	*Production based*	*Cost based*	*Benefits Transfer*
PROVISIONING					
Food *(e.g. Production of fish, wild game/hunting, fruits and grains)*	***Contingent Ranking*** *(e.g Lynam et al., 1994;)* ***CVM*** *(e.g. Gunawardena et al.,1999; Shaikh et al,2007; Loomis 1992)*	***Hedonic pricing*** *(e.g. Livengood 1983; Loomis 1992)* ***Market price*** *(e.g. Pattanayak and Kramer 2001; Chopra and Kadekodi 1997; Moskowitz and Talberth 1998; Verma 2008)* ***TCM*** *(e.g. Barnhill 1999; Loomis 1992)*	***Factor Income*** *(e.g. Peters et al. 1989; Hodgson and Dixon 1998; Carret and Loyer, 2003; Anderson 1987; Mäler 1992)*	***Avoided cost*** *(e.g. Bann, 1999)* ***Mitigation cost*** ***- External cost*** *(Emerton 1999; Madhusudan 2003)* ***Opportunity cost*** *(e.g. Dixon & Sherman 1990; Hodgson & Dixon 1988; Kramer et al. 1992, 1995; Loomis et al. 1989; Ruitenbeek 1989a, 1989b; Emerton 1999)* ***Replacement cost*** *(e.g. Rodriguez et al. 2006)*	***Benefits Transfer*** *(e.g. Costanza et al. 1997)*
Water *(e.g. Storage and retention of water for domestic, industrial and agricultural use*	***CVM*** *(e.g. Sutherland and Walsh 1985;)*	***TCM*** *(e.g. Wittington et al. 1990, 1991)*	***Factor Income*** *(e.g Kumari 1999; Dunkiel and Sugarman 1998)* ***Production Function*** *(e.g. Aylward et al. 1999; Kumari 1996;Wilson & Carpenter 1999; Sedell et al. 2000)*	***Avoided cost*** *(e.g. Chaturvedi, 1993;)* ***Treatment/Mitigation cost*** *(e.g Kumari 1996)*	
Raw Materials *(e.g. fibres, timber, fuelwood, fodder, peat, fertilizer, construction material etc.)*	***Contingent Ranking*** *(e.g. Emerton 1996)* ***CVM*** *(e.g. Kramer et al. 1992, 1995; Shaikh et al. 2007; Olsen and Lundhede 2005)* ***Multi-criteria analysis*** *(e.g. Chopra and Kadekodi 1997)*	***Market prices*** *(e.g. Croitoru 2006; Ammour et al. 2000; Jonish 1992; Sedjo 1988; Sedjo and Bowes, 1991; Veríssimo et al. 1992; Uhl et al. 1992; Verma 2000; Verma 2008)* ***Net Price Method*** *(e.g. Parikh & Haripriya 1998)* ***Substitute Goods*** *(e.g. Adger et al. 1995; Gu-natilake et al. 1993; Chopra 1993; Fleming 1981, cited in Dixon et al. 1994)*	***Factor Income*** *(e.g. Anderson 1987; Peters et al. 1989; Alcorn 1989; Anderson and Jardim 1989; Godoy and Feaw, 1989; Howard 1995;Peters et al. 1989; Pearce 1991; Pinedo-Vasquez et al. 1992; Ruitenbeek 1989a, 1989b; Aakerlund 2000; Kumar and Chopra 2004; Verma, 2008)*	***Opportunity cost*** *(e.g. Chopra et al., 1990; Grieg-Gran, M. 2006; Kramer, R.A., N.P. Sharma, et al. (1995; Niskanen 1998; Emerton (1999; Butry, D.T. and S.K. Pattanayak, 2001; Saastamoinen, 1992; Browder et al. 1996)* ***Replacement Cost*** *(e.g. Ammour et al. 2000)*	

FOREST SERVICES	*Stated Preference*	*Revealed Preference*	*Production based*	*Cost based*	*Benefits Transfer*
Genetic resources *(e.g. biochemichal production models and test-organisms, genes for resistance to plant pathogens)*					
Medicinal resources *(e.g extraction of medicines and other materials from biota)*		***Market price*** *(e.g. Mendelsohn, and Ballick, 1995; Kumar, 2004)*		***Replacement Cost- Forest Rehabilitation*** *(e.g. Cavatassi 2004)*	
Ornamental resources species *(e.g aquarium fish and plants like lotus)*					
Human Habitat *(e.g. forests provide houding to many dwellers)*					
Transport *(e.g Wetlands are source of navigation)*					
REGULATING					
Air quality regulation *(e.g., capturing dust particles*	***Existence + bequest value*** *(e.g. Haefele et al. 1992)*			***Market price / Avoided cost*** *(e.g. Novak et al. 2006; Haefele et al. 1992)* ***Replacement cost*** *(e.g. McPherson 1992; Dwyer et al. 1992;)*	
Climate regulation *(e.g. Source of and sink for greenhouse gases; influence local and regional temperature, precipitation, and other climatic processes incl. Carbon sequestration)*		***Market price*** *(e.g. Clinch ,1999; Loomis and Richardson, 2000; Verma, 2008)*		***Avoided cost*** *(e.g. van Kooten & Sohngen 2007; Dunkiel & Sugarman 1998; Pearce 1994; Turner et al. 2003; Kadekodi & Ravin-dranath 1997; McPherson 1992; Dwyer et al. 1992; Pimentel et al. 1997)* ***Damage Cost*** *(e.g. Howard, 1995)* ***Mitigation Cost*** *(e.g Van Kooten & Sohngen 2007)*	***Benefits transfer*** *(e.g. Dunkiel and Sugarman 1998; Loomis and Richardson 2000;)*

FOREST SERVICES	***Stated Preference***	***Revealed Preference***	***Production based***	***Cost based***	***Benefits Transfer***
				Replacement Cost *(e.g. Howard 1995)*	
Moderation of extreme events *(e.g. storm protection, flood prevention, coastal protection, fire prevention)*	***CVM*** *(e.g. Loomis et al. 1996)*		***Factor Income*** *(e.g. Anderson,1987)*	***Avoided cost*** *(e.g. Pattanayak & Kramer 2001; Loomis & Gonzalez 1997; Yaron 2001; Ruitenbeek 1992; Paris and Ruzicka 1991; Myers 1996)* ***Replacement Cost*** *(e.g. Bann 1998)*	
Regulation of water flows/ Hydrological regimes *(natural drainage, floodplain function, water storage for agriculture or industry, drought prevention, groundwater recharge/ discharge)*		***Public Investments*** *(e.g. Ferraro 2002)*	***Factor income*** *(e.g. Pattanayak & Kramer 2001)*	***Damage cost*** *(e.g. Yaron 2001;)* ***Replacement cost*** *(e.g. Niskanen 1998; McPherson 1992; Dwyer et al. 1992;)*	
Water purification/ detoxification, waste treatment/pollution control *(e.g. retention, recovery, and removal of excess nutrients and other pollutants)*		***TCM*** *(e.g. Wittington et al. 1990, 1991)*		***Restoration cost*** *(e.g. Adger et al. 1995 Mexico)*	
Erosion prevention *(e.g. retention of soils and sediments)*				***Avoided costs*** *(e.g Bann 1999; Paris and Ruzicka 1991)* ***Replacement costs*** *(e.g. Ammour et al. 2000; Kumar 2000)*	
Soil formation /conservation *(e.g.sediment retention and accumulation of organic matter) Note: should come under support services*	***CVM*** *(e.g. Rodriguez et al. 2006;)*			***Avoided cost*** *(e.g; Paris & Ruzicka 1991)* ***Reduced cost of alternate technology cost*** *(e.g. Kadekodi 1997)* ***Replacement cost*** *(e.g. Bann 1998; Ammour et al.*	

FOREST SERVICES	***Stated Preference***	***Revealed Preference***	***Production based***	***Cost based***	***Benefits Transfer***
				2000)	
Pollination *(e.g. habitat for pollinators)*	*;*		***Factor Income*** *(e.g. Ricketts 2004; Pattanayak & Kramer 2001)*	***Replacement cost*** *(e.g. Moskowitz & Talberth 1998)*	
Biological control *(e.g. seed dispersal, pest species and disease control)*	***Option value*** *(e.g. Walsh et al. 1984)* ***Existence + bequest value*** *(e.g. Walsh et al. 1984)*			***Damage cost*** *(e.g. Moskowitz and Talberth 1998; Reid 1999)* ***Replacement cost*** *(e.g. Rodriguez et al. 2006)*	
HABITAT/SUPPORT					
Biodiversity and Nursery service *(e.g. habitats for resident or transient species)*	***Choice Modeling*** *(e.g. Adamowicz et al. 1998b; Hanley et al. 1998;)* ***CVM*** *(e.g. Duffield 1992; Loomis and Ekstrand 1997; Rubin et al. 1991; Loomis et al. 1994; Hagen et al. 1992)*			***Opportunity cost*** *(e.g. Howard 1997;)* ***Replacement cost*** *(e.g. Rodriguez et al. 2006)*	
Gene pool protection/ endangered species proetction		***Public Investments*** *(e.g. Siikamaki & Layton 2007; Burner et al. 2003; Strange et al. 2006; Polasky et al. 2001; Ando et al. 1998)*			
Nutrient cycling *(e.g. storage, recycling, processing, and acqui-sition of nutrients)*					
CULTURAL					
Aesthetic *(e.g. appreciation of natural scenery, other than through deliberate recreational activities)*		***Hedonic pricing*** *(e.g. Garrod & Willis 1992; Tyrvaninen & Meittinen 2000; Kramer et al. 2003; Holmes 1997)* ***TCM*** *(e.g. Holmes 1997)*		***Restoration Cost*** *(e.g Reeves et al. 1999;)*	
Recreation &	***Choice Models***	***TCM***	***Production Function/Factor***		***Benefits transfer***

FOREST SERVICES	*Stated Preference*	*Revealed Preference*	*Production based*	*Cost based*	*Benefits Transfer*
tourism/Ecotourism Wilderness (remote-non-use) *(e.g. Opportunities for tourism and recreational activities)*	*(e.g Adamowicz et al. 1994; Boxall et al. 1996;)* ***CVM*** *(e.g. Adger et al. 1995; Dixon & Sherman 1990; Hadker et al. 1997; Kumari 1995a; Gunawardena et al. 1999; Flatley & Bennett, 1996; Mill et al. 2007; Bateman & Langford 1997; Willis et al. 1998; Bateman et al. 1996; Hanley 1989; Hanley & Ruffell 1991; Hanley & Ruffell 1992; Whinteman & Sinclair 1994; Guruluk 2006; Brown 1992; Sutherland & Walsh 1985; Moskowitz & Talberth 1998; Gilbert et al. 1992; Walsh et al. 1984; Clayton & Mendelsohn 1993; Walsh & Loomis 1989; Champ et al. 1997; Loomis & Richardson* ***Participatory Method*** *(e.g., McDaniels and Roessler, 1998)* ***Option value*** *(e.g. Walsh et al. 1984)*	*(e.g. Tobias and Mendelsohn, 1991; Loomis 1992; Adger et al. 1995; Kramer et al. 1995; Willis et al. 1998; Zandersen 1997; Chopra 1998; Moskowitz and Talberth 1998; Hadker et al. 1995; Van Beukering et al. 2003; Manoharan, 1996; Manoharan and Dutt 1999; Elasser, 1999; Loomis and Ekstrand, 1998; Van der Heide et al. 2005; McDaniels and Roessler 1998; Brown 1992; Loomis and Richardson 2000; Yuan and Christensen 1992; Power 1992; Barnhill 1999; Verma, 2008)*	***Income*** *(e.g. Hodgson and Dixon 1988;*		*Walsh and Loomis 1989; Zandersen et al., 2007, 2009.*
Educational *(e.g. Opportunities for formal and informal education and training)*		***TCM*** *(e.g. Power 1992;)*			
Spiritual & artistic inspi-ration *(e.g. source of inspiration; many religions attach spiritual, scared and religious values to aspects of wetland and forest ecosystems)*	***Deliberative monetary valuation*** *(e.g. Hanley et al., 2002)* ***Contingent Ranking*** *(e.g. Garrod and Willis 1997)* ***CVM*** *(e.g. Maharana et. al., 2000)* ***CVM / Choice Modelling** (e.g. Aakerlund, 2000; Mill et al., 2007; Kniivila, M., V. Ovaskainen, et al., 2002;*	***TCM*** *(e.g. Maharana et. al., 2000)*			

FOREST SERVICES	***Stated Preference***	***Revealed Preference***	***Production based***	***Cost based***	***Benefits Transfer***
	McDaniels and Roessler, 1998; Maharana et. Al., 2000)				
Cultural heritage and identity *(e.g. sense of place and belonging)*					
Information for cognitive development					

Table A2.a **Conceptual matrix based on wetland ecosystem services and valuation approaches**

SERVICES	Wetlands				
	Stated Preference	***Revealed Preference***	***Production based***	***Cost based***	***Benefits Transfer***
PROVISIONING	***Choice modelling*** *Layton et al. 1998; Seferlis 2004; Psychoudakis et al. 2004; Carlsson et al. 2003; Gordon et al. 2001;)* ***Contingent ranking*** *(e.g. Emerton 1996;)* ***CVM*** *(e.g. Bergstrom 1990; Costanza et al. 1997; Hammack & Brown 1974; Benessaiah 1998; Bhatta 2000; Hanley& Craig 1991)* ***CVM for non-user benefits*** *(e.g. James and Murty, 1999;)* ***Participatory Valuation*** *(e.g. Eaton, 1997; Emerton, 2005; IUCN-WANI, 2005;)*	***Public Investments*** *(e.g Powicki 1998 ; Emerton, 2005)*	***Bio-economic Modelling*** *(e.g. Hammack& Brown 1974)* ***Factor Income/Production Function*** *(e.g. Barbier et al. 1991; Barbier et. al. 1993; Hammack and Brown 1974; Costanza et al., 1997; Hodgson & Dixon 1988; Emerton 1998; Bann 1999; Gammage 1997;Barbier and Strand 1998; Janssen and Padilla 1997; Nickerson 1999; Verma et al. 2003; Khalil 1999; Emerton, 2005; Stuip et al. 2002; Benessaiah 1998; Ruitenbeek 1994; Verma et al. 2003; Emerton, 2005; Vidanage et al. 2005)*	***Avoided cost*** *(e.g. L.Emerton, 2005)* ***Conversion Cost*** *(e.g. R. Abila,1998)* ***Public Investments*** *(e.g Powicki 1998 ; L.Emerton, 2005)* ***Opportunity cost*** *(e.g. Dixon and Sherman 1990; Hodgson and Dixon, 1988; Kramer et al.1992, 1995; L.Emerton, 2005, Ruitenbeek, 1989a, 1989b)* ***Replacement cost*** *(e.g. Grenet al. 1994; Abila 1998)* ***Restoration cost*** *(e.g. Verma et al. 2003; Emerton 2005)*	***Benefits Transfer*** *(e.g.White et al. 2000; Stuip et al. 2002; Costanza et al., 1997 ; Schuijt 2002; Seidl and Moraes 2000 ; White et al. 2000)*
REGULATING	***Choice modelling*** *(e.g. Adamowicz et al. 1994; Birol et al. 2007; Ragkos et al. 2006; Colombo et al. 2004; Colombo et al. 2006;)* ***CVM*** *(e.g. Hanley & Craig 1991; Bateman et al. 1993; Gren, 1995; Loomis, 2000)* ***Participatory Valuation*** *(e.g Emerton, 2005; IUCN-WANI, 2005)*		***Production function/ Factor Income*** *(e.g. Acharya, 2000; Acharya and Barbier 2000; Gren, 1995; Seidl, 2000)*	***Avoided cost*** *(e.g. Emerton 1998; Emerton 2005; Emerton 2003; Bann 1999; Verma et al. 2003)* ***Mitigation Cost*** *(e.g. Sankar 2000)* ***Replacement Cost*** *(e.g Gupta 1975; Farber 1987; Gren et al. 1994; Emer-ton 2005; Gren et al. 1994; IUCN 2003; Stuip et al. 2002)* ***Restoration cost*** *(e.g. Emerton 2005; Gren, 1995; Verma et al., 2003)*	***Benefits Transfer*** *(e.g. Costanza et.al., 1997; Seidl and Moraes 2000)*
HABITAT/SUPPORT	***Choice modeling*** *(e.g. Brouwer et al. 2003)* ***CVM***		***Production function/ Factor Income*** *(e.g. Barbier and Thompson*	***Replacement cost*** *(e.g. Gren et al. 1994)*	***Benefits Transfer*** *(e.g. Andréassen-Gren & Groth*

SERVICES	Wetlands				
	Stated Preference	***Revealed Preference***	***Production based***	***Cost based***	***Benefits Transfer***
	(e.g. Eija Moisseinen 1993; Ragos et al. 2006)		*1998; Johnston 2002; Lynne et al. 1981; Ramdial 1975*)		*1995; White et al. 2000)*
CULTURAL	***Choice modelling*** *(e.g. Bergland 1997; Tuan et al. 2007; Boxall et al. 1996; Carlsson et al. 2003; Hanley et al. 2002; Horne et al. 2005; Boxall and Adamomicz 2002; Adamowicz et al. 1994; Adamowicz et al. 1998b; Pak and Turker, 2006)* ***CVM*** *(e.g. Mahan, B.L., 1997; Thibodeu & Ostro 1981; Naylor & Drew 1998; Murthy & Menkhuas, 1994; Manoharan, 1996; Costanza et al., 1997; Manoharan and Dutt, 1999; Maharana et. al. 2000; Wilson & Carpenter 2000; Stuip et al. 2002; Bergstrom, 1990; W.Bell 1996; Shultz et al. 1998; Tuan et al. 2007)* ***Participatory Valuation*** *(e.g. IUCN-WANI, 2005)*	***Consumer Surplus*** *(e.g. Bergstrom et al. 1990)* ***Hedonic pricing*** *(e.g. Verma et al. 2003; Mahan 1997)* ***TCM*** *(e.g. Farber 1987; Willis et al. 1991Chopra 1998; Hadker et al. 1995; Manoharan, 1996; Pak and Turker, 2006)*	***Production function/ Factor Income*** (e.g. Costanza et al. 1989)	***Opportunity Cost*** *(e.g. Loomis et al. 1989;)* ***Protection cost*** *(e.g. Pendleton 1995)* ***Replacement Cost*** *(e.g. Abila,1998; Gupta, 1975)*	***Benefits Transfer*** *(e.g. M. Andréassen-Gren & K.H. Groth, 1995; Sorg and Loomis 1984;Walsh et al. 1988; MacNair 1993; Loomis et al. 1999; Markowski et al. 1997; Rosenberger and Loomis 2000; Seidl and Moraes 2000; White et al. 2000)*

Table A2.b **Conceptual matrix based on forest ecosystem services and valuation approaches**

SERVICES	**Forest**				
	Stated Preference	***Revealed Preference***	***Production based***	***Cost based***	***Benefits Transfer***
PROVISIONING	***Contingent Ranking*** *(e.g Lynam et al., 1994;. Emerton,1996)* ***CVM*** *(e.g. Gunawardena et al.,1999; Shaikh et al,2007; Kramer et al., 1992, 1995; Olsen and Lundhede, 2005; Loomis 1992; Sutherland and Walsh 1985)* ***Existence + bequest value*** *(e.g. Haefele et al. 1992)* ***Multi-criteria analysis*** *(e.g. Chopra and Kadekodi, 1997)*	***Hedonic pricing*** *(e.g. Livengood 1983; Loomis 1992)* ***Market price*** *(e.g. Pattanayak & Kramer 2001; Croitoru 2006; Ammour et al. 2000; Chopra & Kadekodi 1997; Moskowitz & Talberth 1998; Jonish 1992; Sedjo 1988; Sedjo & Bowes 1991; Veríssimo et al. 1992; Verma 2000; Verma 2008; Mendelsohn & Ballick 1995; Kumar 2004; Uhl et al. 1992)* ***Net Price Method*** *(e.g. Parikh &Haripriya 1998)* ***Substitute Goods*** *(e.g. Adger et al.1995; Gunatilake et al.1993; Chopra,1993; Fleming 1981, cited in Dixon et al. 1994)* ***TCM*** *(e.g.Wittington et al. 1990, 1991; Barnhill 1999; Loomis 1992)*	***Factor Income*** *(e.g Kumari ,1999; Dunkiel and Sugarman, 1998; . Peters et al.,1989;Hodgson and Dixon,1998; Carret and Loyer, 2003; Anderson 1987; Maler 1992; Anderson 1987; Peters et.al.,1989;Alcorn,1989; Anderson and Jardim,1989; Godoy and Feaw, 1989; Howard 1995;Peters et al.,1989; Pearce,1991; Pinedo-Vasquez et al.,1992; Ruitenbeek,1989a, 1989b;Aakerlund,2000; Verma, 2008)* ***Production Function*** *(e.g Aylward et al. 1999; Kumari 1996;Wilson and Carpenter 1999;Sedell et al. 2000; Kumar and Chopra, 2004)*	***Avoided cost*** *(e.g Bann, 1999; Chaturvedi, 1993)* ***Mitigation cost*** *(e.g. Emerton 1999; Madhusudan 2003)* ***Opportunity cost*** *(e.g. Dixon & Sherman 1990; Hodgson & Dixon 1988; Kramer et al. 1992 1995; Loomis et al. 1989; Ruitenbeek 1989a, 1989b; Emerton 1999; Chopra et al. 1990; Grieg-Gran 2006; Kramer et al. 1995; Niskanen 1998; Emerton 1999; Butry & Pattanayak 2001; Saastamoinen 1992; Browder et al. 1996)* ***Rehabilitation cost*** *(e.g. Cavatassi 2004)* ***Replacement Cost*** *(e.g. Ammour et al. 2000**;** Rodriguez et al. 2006;)* ***Treatment/Mitigation cost*** *(e.g Kumari 1996)*	***Benefits transfer*** *Costanza et.al., 1997)*
REGULATING	***CVM*** *(e.g. Loomis J.B., C.A. Gonzale & R. Gregory, 1996); Rodriguez et al. 2006;)* ***Option value*** *(e.g. Walsh et al. 1984)*	***Market price*** *(e.g. Clinch ,1999; Loomis and Richardson, 2000; Verma, 2008)* ***Public Investments*** *(e.g. Ferraro, P.J. , 2002)* ***TCM*** *(e.g. Wittington et al. 1990, 1991*	***Factor Income*** *(e.g. Anderson,1987;. Pattanayak and Kramer, 2001; Ricketts 2004)*	***Avoided cost*** *(e.g. Novak et al. 2006; Haefele et al. 1992; van Kooten & Sohngen 2007; Dunkiel & Sugarman 1998; Pearce 1994; Turner et al. 2003; Kadekodi & Ravindranath 1997; Bann 1999; Paris & Ruzicka 1991; McPherson 1992; Dwyer et al. 1992; Pimentel et al. 1997; Myers 1996)* ***Damage Cost*** *(e.g. Howard 1995; Yaron 2001;*	***Benefits transfer*** *(e.g. Dunkiel and Sugarman 1998; Loomis and Richardson 2000)*

SERVICES	Forest				
	Stated Preference	***Revealed Preference***	***Production based***	***Cost based***	***Benefits Transfer***
				Moskowitz & Talberth 1998; Reid 1999) ***Mitigation Cost*** *(e.g Van Kooten & Sohngen 2007)* ***Reduced cost of alternate technology cost*** *(e.g. Kadekodi 1997)* ***Restoration cost*** *(e.g. Adger et al. 1995)* ***Replacement Cost*** *(e.g. Howard 1995; Ammour et al. 2000; Kumar 2000; McPherson 1992; Dwyer et al. 1992; Moskowitz & Talberth 1998; Rodriguez et al. 2006)*	
HABITAT/ SUPPORT	***Choice Modeling*** *(e.g. Adamowicz et al. 1998b; Hanley et al., 1998)* ***CVM*** *(e.g. Duffield 1992; Loomis & Ek-strand 1997; Rubin et al. 1991; Loomis et al. 1994; Hagen et al. 1992)*	***Public Investments*** *(e.g. Siikamaki and Layton, 2007; Burner et al., 2003; Strange et al., 2006; Polasky et al. 2001; Ando et al,1998)*		***Opportunity cost*** *(e.g. Howard 1997)* ***Replacement cost*** *(e.g. Rodriguez et al. 2006)*	
CULTURAL	***Choice Modelling*** *(e.g. Aakerlund, 2000; Mill et al., 2007; Kniivila, M., V. Ovaskainen, et al., 2002; McDaniels and Roessler, 1998; (Maharana et al., 2000)* ***Contingent Ranking*** *(e.g. Garrod and Willis 1997)* ***CVM*** *(e.g. Maharana et. al., 2000 ; Brown 1992; Sutherland and Walsh 1985; Moskowitz and Talberth 1998; Gilbert et al. 1992; Walsh et al. 1984; Clayton and Mendelsohn 1993; Walsh and Loomis 1989; Champ et al. 1997;*	***Hedonic pricing*** *(e.g. Garrod & Willis 1992; Tyrvaninen & Meittinen 2000; Kramer et al. 2003)* ***TCM*** *(e.g. Tobias & Mendelsohn 1991; Loomis 1992; Adger et al. 1995; Kramer et al. 1995; Willis et al. 1998; Zandersen 1997, Chopra 1998; Moskowitz & Talberth 1998; Hadker et al. 1995; Van Beukering et al. 2003; Manoha-ran 1996; Manoharan & Dutt 1999; Elasser 1999; Loomis & Ekstrand 1998; Van der Heide et*	***Production Function/Factor Income*** *(e.g. Hodgson and Dixon 1988)*	***Restoration Cost*** *(e.g. Reeves et al. ,1999)*	***Benefits transfer*** *(e.g. Walsh and Loomis 1989; Zandersen et al., 2007, 2009)*

SERVICES	Forest				
	Stated Preference	***Revealed Preference***	***Production based***	***Cost based***	***Benefits Transfer***
	Loomis and Richardson, 2000; Verma, 2008) ***Deliberative monetary valuation*** *(e.g. Hanley et al., 2002)* ***Option value*** *(e.g. Walsh et al. 1984)*	*al. 2005; McDaniels & Roessler 1998; Maharana et al. 2000; Holmes 1997; Power 1992; Brown 1992; Loomis & Richard-son 2000; Yuan & Christensen 1992; Power1992; Barnhill1999)*			

Table A3. Matrix linking specific value types, valuation methods and ecosystem services – Examples from wetland and forest ecosystems

Note: NA = Not Applicable i.e. particular combination of value type and use is unlikely (based on TEV+ MA classification amalgamation matrix)

	SERVICES	**Wetlands**				**Forests**			
		Direct use	**Indirect use**	**Option use**	**Non-use**	**Direct use**	**Indirect use**	**Option use**	**Non-use**
	PROVISIONING								
1	***Food*** *(e.g. Production of fish, wild game/ hunting, fruits and grains)*	***<u>Stated preference</u>*** ***Choice modelling*** *(e.g. Layton et al. 1998; Seferlis 2004; Psychoudakis et al. 2004; Carlsson et al. 2003)* ***Contingent ranking*** *(e.g. Emerton 1996)* ***CVM*** *(e.g. Bergstrom, 1990; Costanza et.al., 1997; Hammack & Brown 1974; Benessaiah 1998)* ***Participatory Valuation*** *(e.g. Emerton, 2005; IUCN-WANI, 2005;)* ***<u>Production based</u>*** ***Factor Income/Production Function*** *(e.g. Barbier, Adams & Kimmage 1991; Barbier et. al., 1993; Hammack & Brown, 1974; Costanza et al. 1997; Hodgson & Dixon 1988; Emerton 1998; Bann 1999; Gammage 1997; Barbier & Strand 1998; Janssen & Padilla 1997; Nickerson 1999; Verma*	***NA***	***<u>Stated preference</u>*** ***CVM*** *(e.g. Costanza et.al., 1997)* ***Stakeholder Analysis and CVM*** *(e.g. Bhatta, 2000)* ***<u>Cost based</u>*** ***Restoration cost*** *(e.g. Emerton, 2005)*	***NA***	***<u>Stated preference</u>*** ***Contingent Ranking*** *(e.g Lynam et al. 1994)* ***CVM*** *(e.g. Gunawar-dena et al. 1999; Shaikh et al. 2007; Loomis 1992)* ***<u>Revealed preference</u>*** ***Hedonic pricing*** *(e.g. Livengood 1983; Loomis 1992)* ***Market price*** *(e.g. Pattanayak & Kramer 2001; Chopra & Kadekoid 1997; Verma 2008)* ***TCM*** *(Barnhill 1999; Loomis 1992)* ***<u>Production based</u>*** ***Factor Income*** *(e.g. Peters et al. 1989; Hodgson & Dixon 1998; Carret & Loyer 2003; Anderson 1987; Mäler 1992; Moskowitz &Talberth 1998; Verma, 2008)* ***<u>Cost based</u>*** ***Mitigation cost*** ***-External cost*** *(e.g. Emerton 1999; Madhusudan 2003)*	***NA***		***NA***

	SERVICES	Wetlands				Forests			
		Direct use	**Indirect use**	**Option use**	**Non-use**	**Direct use**	**Indirect use**	**Option use**	**Non-use**
		2001; Khalil 1999; Emerton 2005; Stuip et al. 2002; Benessaian 1998) ***<u>Cost based</u>*** ***Replacement cost*** *(e.g. Gren, et al. 1994; Abila 1998)* ***<u>Benefits transfer</u>*** ***Benefits Transfer*** *(e.g.White et al. 2000; Stuip et al. 2002)*				***Net Revenue*** ***Avoided cost*** *(e.g Bann, 1999)* ***Opportunity cost*** *(e.g. Dixon & Sherman 1990; Hodgson & Dixon 1988; Kramer et al. 1992, 1995; Loomis et al. 1989; Ruitenbeek, 1989a, 1989b; Emerton 1999)* ***Replacement cost*** *(e.g. Rodriguez et al. 2006;)*			
2	***Water*** *(e.g. Storage and retention of water for domestic, industrial and agricultural use)*	***<u>Stated preference</u>*** ***Choice modelling*** *(e.g. Gordon et al. 2001)* ***Participatory Valuation*** *(e.g. IUCN-WANI 2000)* ***<u>Revealed preference</u>*** ***Public Investments*** *(e.g Powicki 1998 ; Emerton, 2005)* ***<u>Production based</u>*** ***Factor Income*** *(e.g. Emerton, 2005)* ***<u>Cost based</u>*** ***Opportunity cost*** *(e.g. Emerton 2005)* ***Replacement cost*** *(e.g. Gren et al., 1994)* ***Restoration cost*** *(e.g. Verma, 2001)*	***NA***	***<u>Cost based</u>*** ***Restoration cost*** *(e.g. L.Emerton, 2005)*	***NA*** ***CVM for non-user benefits*** *(e.g. James and Murty, 1999;*	***<u>Revealed preference</u>*** ***TCM*** *(e.g. Wittington et al. 1990, 1991)* ***<u>Production based</u>*** ***Factor Income*** *(e.g Kumari , 1999; Dunkiel & Sugarman 1998)* ***Production Function*** *(e.g. Aylward et al. 1999; Kumari 1996; Wilson & Carpenter 1999; Sedell et al. 2000)* ***<u>Cost based</u>*** ***Avoided cost*** *(e.g. Chaturvedi, 1993)* ***Treatment/ Mitigation cost*** *(e.g Kumari 1996)*	***NA***	***<u>Stated preference</u>*** ***CVM*** *(e.g. Kadekodi, 2000;)*	***NA***
3	***Raw Materials****(e.g. fibres, timber, fuelwood, fodder, peat,fertilizer, construction material etc.)*	***<u>Stated preference</u>*** ***Contingent ranking*** *(e.g. Emerton ,1996;)* ***CVM*** *(e.g. Hanley & Craig 1991)* ***Participatory Valuation*** *(e.g. Eaton, 1997; Emerton, 2005; IUCN-WANI, 2005;)*	***NA***	***<u>Stated preference</u>*** ***Participatory valuation*** *(e.g. Eaton, 1997)* ***<u>Cost based</u>*** ***Restoration cost*** *(e.g. Emerton, 2005)*	***NA***	***<u>Stated preference</u>*** ***Contingent Ranking*** *(e.g. Emerton 1996;)* ***CVM*** *(e.g. Kramer et al. 1992, 1995; Shaikh et al., 2007; Olsen and Lundhede 2005)* ***Multi-criteria analysis***	***NA***	***<u>Stated preference</u>*** ***CVM*** *(e.g. Ninan and Sathyapalan, 2005;)* ***<u>Cost based</u>*** ***Shadow price*** *(e.g Godoy and*	***NA***

	SERVICES	Wetlands				Forests			
		Direct use	**Indirect use**	**Option use**	**Non-use**	**Direct use**	**Indirect use**	**Option use**	**Non-use**
		Production based ***Factor Income*** *(e.g. Halil, 1999; Ruitenbeek 1994; Verma, 2001; Emerton, 2005; Stuip et al. 2002)* ***Cost based*** ***Opportunity cost*** *(e.g. Dixon and Sherman 1990; Hodgson and Dixon, 1988; Kramer et al.1992, 1995; L.Emerton, 2005, Ruitenbeek, 1989a, 1989b)* ***Replacement cost*** *(e.g Gren et al. 1994)*				*(e.g. Chopra & Kadekodi 1997)* ***Revealed preference*** ***Market prices*** *(e.g.Croitoru 2006; Ammour et al. 2000 ; Jonish, 1992; Sedjo, 1988; Sedjo and Bowes 1991; Veríssimo et al. 1992; Verma, 2000; Verma, 2008 Uhl et al., 1992)* ***Net Price Method*** *(e.g. Parikh and Haripriya 1998)* ***Substitute Goods*** *(e.g. Adger et al. 1995; Gunatilake et al. 1993; Chopra, 1993; Fleming 1981, cited in Dixon et al. 1994)* ***Production based*** ***Factor Income*** *(e.g. Anderson 1987; Peters et al. 1989; Alcorn 1989; Anderson & Jardim 1989; Godoy & Feaw, 1989; Howard 1995; Peters et al. 1989; Pearce 1991; Pinedo-Vasquez et al. 1992; Ruiten-beek 1989a, 1989b; Aakerlund 2000; Kumar & Chopra 2004; Verma 2008)* ***Cost based*** ***Opportunity cost*** *(e.g. Chopra et al. 1990; Grieg-Gran 2006; Kramer, Sharma et al. 1995; Niskanen 1998; Emerton 1999; Butry, Pattanayak, 2001; Saastamoinen, 1992; Browder et al. 1996)* ***Replacement Cost***		*Feaw 1989)*	

	SERVICES	Wetlands				Forests			
		Direct use	Indirect use	Option use	Non-use	Direct use	Indirect use	Option use	Non-use
						(e.g. Ammour et al. 2000)			
4	***Genetic resources*** *(e.g. biochemichal production models and test-organisms, genes for resistance to plant pathogens;)*	***Stated preference*** **Participatory Valuation** *(e.g. Emerton, 2005)* ***Production based*** **Bioeconomic Modelling** *(e.g. Hammack and Brown, 1974)*	***NA***		***NA***		***NA***	***Stated preference*** ***CVM*** *(e.g Veistern et al., 2003)*	***NA***
5	***Medicinal resources*** *(e.g extraction of medicines and other materials from biota)*	***Stated preference*** **Participatory Valuation** *(e.g. Emerton, 2005; IUCN-WANI, 2005)* ***Cost based*** **Avoided cost** *(e.g. Emerton, 2005)*	***NA***	***Cost based*** **Restoration cost** *(e.g. Emerton, 2005)*	***NA***	***Revealed Preference*** **Market Price** *(e.g. Mendelsohn, & Ballick 1995; Kumar 2004)* ***Cost based*** **Replacement Cost-Forest Rehabilitation** *(e.g. Cavatassi, 2004)*	***NA***		***NA***
6	***Ornamental resources species*** (e.g aquarium fish and plants like lotus)	***Stated preference*** **Participatory Valuation** *(e.g. Emerton, 2005)* ***Production based*** **Factor Income** *(e.g. Vidanage et al. 2005)*	***NA***		***NA***		***NA***		***NA***
	Human Habitat *(e.g. forest provide houding to many dwellers)*		***NA***		***NA***		***NA***		***NA***
	Transport *(e.g Wetlands are source of navigation)*	***Cost based*** **Conversion Cost** *(e.g. Abila,1998)*	***NA***		***NA***		***NA***		***NA***
	REGULATING								
7	***Air quality regulation*** *(e.g., capturing dust particles*	***NA***			***NA***	***NA***	***Cost based*** **Market price / Avoided cost** *(e.g. Novak et al., 2006; Haefele et al.*		***NA*** ***Existence + bequest value*** *Haefele et al. 1992*

	SERVICES	Wetlands				Forests			
		Direct use	**Indirect use**	**Option use**	**Non-use**	**Direct use**	**Indirect use**	**Option use**	**Non-use**
							1992) ***Replacement cost*** *(e.g. McPherson 1992; Dwyer et al. 1992;)*		
8	***Climate regulation*** *(e.g. Source of and sink for greenhouse gases; influence local and regional temperature, precipitation, and other climatic processes incl. Carbon sequestration)*	***NA***	***Stated preference*** ***Participatory Valuation*** *(e.g. Emerton, 2005)* ***Cost based*** ***Avoided cost*** *(e.g. Emerton, 1998; Emerton, 2003)*		***NA***	***NA***	***Revealed preference*** ***Market price*** *(e.g. Clinch , 1999; Loomis and Richardson; 2000; Verma, 2008)* ***Cost based*** ***Avoided cost*** *(e.g. van Kooten & Sohngen 2007; Dunkiel & Sugarman 1998; Pearce,1994; Turner et al. 2003; Kadekodi & Ravindranath, 1997; McPherson 1992; Dwyer et al. 1992; Pimentel et al. 1997)* ***Damage Cost*** *(e.g. Howard, 1995)* ***Mitigation Cost*** *(e.g Van Kooten & Sohngen 2007)* ***Replacement Cost*** *(e.g. Howard 1995)* ***Benefits transfer*** ***Benefits transfer*** *(e.g. Dunkiel & Sugarman 1998; Loomis & Richardson 2000)*		***NA***
9	***Moderstion of extreme events*** *(e.g. storm proetction, flood*	***NA***	***Stated preference*** ***CVM*** *(e.g. Hanley and Craig, 1991;*		***NA***	***NA***	***Stated preference*** ***CVM*** *(e.g. Loomis & Gonzalez 1997)*		***NA***

	SERVICES	Wetlands				Forests			
		Direct use	**Indirect use**	**Option use**	**Non-use**	**Direct use**	**Indirect use**	**Option use**	**Non-use**
	prevention, coastal protection, fire prevntion)		*Bateman et al., 1993)* ***Participatory Valuation*** *(e.g. Emerton 2005)* ***Cost based*** ***Avoided cost*** *(e.g. Bann 1999; Costanza et al., 1997)* ***Replacement Cost*** *(e.g. Gupta 1975; Farber, 1987)*				***Production based*** ***Factor Income*** *(e.g. Anderson 1987;)* ***Cost based*** ***Avoided cost*** *(e.g. Pattanayak & Kramer 2001; Loomis & Gonzalez 1997; Yaron 2001; Ruiten-beek, 1992; Paris & Ruzicka 1991; Myers 1996)* ***Replacement Cost*** *(e.g. Bann 1998)*		
10	***Regulation of water flows/ Hydrological regimes*** *(natural drainage, floodplain function, storage of water for agriculture or industry, drought prevention, groundwater recharge/ discharge)*	***NA***	***Stated preference*** ***Choice modelling*** *(e.g. Adamowicz et al. 1994; Birol et al. 2007; Ragkos et al. 2006)* ***Participatory Valuation*** *(e.g. Emerton 2005; IUCN-WANI 2005)* ***Production based*** ***Factor Income*** *(e.g. Acharya 2000)* ***Cost based*** ***Avoided cost*** *(e.g. Emerton 2005)* ***Replacement cost*** *(e.g. Gren et al. 1994)* ***Restoration cost*** *(e.g. Emerton 2005)*		***NA***	***NA***	***Revealed preference*** ***Public Investments*** *(e.g. Ferraro, P.J. , 2002)* ***Production based*** ***Factor income*** *(e.g. Pattanayak and Kramer, 2001)* ***Cost based*** ***Replacement cost*** *(e.g. Niskanen 1998;)*	***PES*** *(e.g. Proano, C.E., 2005).*	***NA***
11	***Water purification/detoxification , and waste***	***NA***	***Stated preference*** ***CVM*** *(e.g. Gren, 1995)*		***NA*** ***CVM*** *(e.g. James*	***NA***	***Revealed preference*** ***TCM*** *(e.g. Wittington et al.*		***NA***

	SERVICES	Wetlands				Forests			
		Direct use	**Indirect use**	**Option use**	**Non-use**	**Direct use**	**Indirect use**	**Option use**	**Non-use**
	treatment/pollution control *(e.g. retention, recovery, and removal of excess nutrients and other pollutants)*		***Production based*** ***Factor Income*** *(e.g. Gren, 1995)* ***Cost based*** ***Avoided costs*** *(e.g Verma, 2001)* ***Mitigation Cost*** *(e.g. Sankar 2000)* ***Restoration cost*** *(e.g. Gren 1995; Verma, 2001)* ***Replacement cost*** *(e.g. Emerton 2005; Gren et al. 1994; IUCN 2003; Stuip et al. 2002)*		*and Murty, 1999)*		*1990, 1991)* ***Cost based*** ***Restoration cost*** *(e.g. Adger et al. 1995 Mexico)*		
12	***Erosion prevention*** *(e.g. retention of soils and sediments)*	***NA***	***Stated preference*** ***CVM*** *(e.g. Hanley & Craig 1991; Bateman et al. 1993; Loomis 2000)* ***Participatory Valuation*** *(e.g .Emerton 2005)*		***NA***	***NA***	***Cost based*** ***Replacement costs /Avoided costs*** *(e.g. Ammour et al., 2000; Kuma , 2000; Bann, 1999; Paris and Ruzicka , 1991)*		***NA***
13	***Soil formation /conservation*** *(e.g.sediment retention and accumulation of organic matter)* *Note: should come under support services*	***NA***	***Stated preference*** ***Choice modelling*** *(e.g. Colombo et al. 2004; Colombo et al. 2006;)* ***CVM*** *(e.g. Loomis, 2000)* ***Cost based*** ***Restoration cost*** *(e.g. Emerton, 2005)*		***NA***	***NA***	***Stated preference*** ***CVM*** *(e.g. Rodriguez et al. 2006;)* ***Cost based*** ***Avoided cost*** *(e.g; Paris and Ruzicka, 1991;)* ***Income factor/ Replacement cost*** *(e.g. Bann, 1998; Ammour et al. 2000)* ***Reduced cost of***		***NA***

	SERVICES	**Wetlands**				**Forests**			
		Direct use	**Indirect use**	**Option use**	**Non-use**	**Direct use**	**Indirect use**	**Option use**	**Non-use**
							alternate technology cost *(e.g. Kadekodi 1997)*		
14	***Pollination*** *(e.g. habitat for pollinators)*	***NA***	***Production based*** ***Factor Income*** *(e.g. Seidl, 2000)*		***NA***	***NA***	***Production based*** ***Factor Income*** *(e.g. Ricketts, 2004; Patta-nayak & Kramer 2000)* ***Cost based*** ***Replacement cost*** *(e.g. Moskowitz and Talberth 1998;)*		***NA***
15	***Biological control*** *(e.g. seed dispersal, pest species and disease control)*	***NA***			***NA***	***NA***	***Cost based*** ***Damage cost*** *Moskowitz &Talberth 1998; Reid 1999* ***Replacement cost*** *Rodriguez et al. 2006;*	***Stated preference*** ***Option value*** *Walsh et al. 1984*	***NA*** ***Existence + bequest value*** *Walsh et al. 1984*
	HABITAT/ SUPPORT								
16	***Biodiversity and Nursery service*** *(e.g. habitats for resident or transient species)*	***Stated preference*** ***Choice modeling*** *(e.g. Brouwer et al. 2003)*		***Cost based*** ***Replacement cost*** *(e.g. Gren, I., Folke, C., Turner, K. and I. Bateman 1994,)*				***Cost based*** ***Opportunity cost*** *(e.g. Howard 1997)* ***Replacement cost*** *(e.g. Rodriguez et al. 2006;)*	***Stated preference*** ***Choice Modeling*** *(e.g. Adamowicz et al. 1998b; Hanley et al., 1998)*
17	***Gene pool protection/ endangered species proetction***			***Stated preference*** ***CVM*** *(e.g. Eija Moisseinen 1993;)* ***Cost based*** ***Replacement cost*** *(e.g. Gren et al. 1994; Bateman 1994)*		***Revealed preference*** ***Public Investments*** *(e.g. Siikamaki and Layton, 2007; Burner et al., 2003; Strange et a.l, 2006; Polasky et al., 2001; Ando et al. 1998)*		***Cost based*** ***Opportunity cost*** *(e.g. Chomitz, Alger, et al., 2005)*	***Stated preference*** ***CVM*** *(e.g. Veistern et al., 2003; Lehtonen et al., 2003; Mallawaarachi et al. 2001;*

	SERVICES	Wetlands				Forests			
		Direct use	**Indirect use**	**Option use**	**Non-use**	**Direct use**	**Indirect use**	**Option use**	**Non-use**
									Garber-Yonts, Kerkvliet et al. 2004)
	Nutrient cycling *(e.g. Storage, recycling, processing, and acquisition of nutrients)*		***Stated preference*** ***Choice Modelling*** *(e.g. Carlsoon et al. 2003)* ***Cost based*** ***Replacement cost*** *(e.g. Gren et al. 1994)* ***Benefits transfer*** ***Benefits Transfer*** *(e.g. Andréassen Gren & Groth, 1995)*						
	CULTURAL								
18	***Aesthetic*** *(e.g. appreciation of natural scenery, other than through deliberate recreational activities)*	***Stated preference*** ***Choice modelling*** *(e.g. Bergland 1997)* ***CVM*** *(e.g. Mahan, 1997)* ***Revealed preference*** ***Hedonic pricing*** *(e.g. Verma 2000; Mahan, 1997)* ***Cost based*** ***Replacement Cost*** *(e.g. Gupta, 1975;)*	***NA***			***Revealed preference*** ***Hedonic pricing*** *(e.g. Garrod & Willis 1992; Tyrvaninen and Meittinen 2000; Kramer et al. 2003; Holmes 1997)* ***TCM*** *(e.g. Holmes 1997)* ***Cost based*** ***Restoration Cost*** *(e.g. Reeves et al. 1999)*	***NA***		
19	***Recreation & tourism/ Ecotou-rism, Wilderness (remote-non-use)*** *(e.g.Opportunities for tourism and recreational activities)*	***Stated preference*** ***Choice modelling*** *(e.g. Boxall et al. 1996; Carlsson et al. 2003; Hanley et al. 2002; Horne et al. 2005; Boxall & Adamomicz 2002; Adamowicz et al. 1994;*	***NA***	***Stated preference*** ***CVM*** *(e.g. Desvousges et al. 1987)*		***Stated preference*** ***Choice Models*** *(e.g Adamowicz et al. 1994; Boxall et al. 1996;)* ***CVM*** *(e.g Adger et al. 1995; Dixon & Sherman 1990; Hadker et al. 1997; Kumari 1995a;*	***NA***	***Stated preference*** ***Option value*** *(e.g. Walsh et al. 1984)* ***Revealed preference*** ***Expenditure on Wilderness***	***Stated preference*** ***Choice Modeling*** *(e.g. Hanley et. al.,1998;)* ***CVM*** *(e.g. Loomis*

	SERVICES	**Wetlands**				**Forests**			
		Direct use	**Indirect use**	**Option use**	**Non-use**	**Direct use**	**Indirect use**	**Option use**	**Non-use**
		Adamowicz et al. 1998b) ***CVM*** *(e.g. Thibodeu & Ostro 1981; Naylor & Drew 1998; Murthy & Menkhuas, 1994; Manoharan, 1996; Costanza et al., 1997; Manoharan & Dutt 1999; Maharana et. al. 2000; Wilson & Carpenter 2000; Stuip et al. 2002; Bergstrom, 1990; Bell 1996; Pak and Turker, 2006)* ***Participatory Valuation*** *(e.g. IUCN-WANI 2005)* ***<u>Revealed preference</u>*** ***Consumer Surplus*** *(e.g. Bergstrom et al. 1990)* ***TCM*** *(e.g. Farber, 1987; Chopra 1998; Hadker et al., 1995; Manoharan, 1996; Pak and Turker, 2006; Willis et. al. 1991)* ***<u>Cost based</u>*** ***Opportunity Cost*** *(e.g. Loomis et al. ,1989;)* ***Protection cost*** *(e.g. Pendleton 1995)* ***Replacement and Conversion Cost*** *(e.g. R. Abila,1998;)* ***<u>Benefits transfer</u>*** ***Benefits Transfer*** *(e.g. Sorg and Loomis 1984; Walsh et al. 1988; MacNair 1993; Loomis et al. 1999; Markowski et al.*				*Gunawardena et al. 1999; Flatley &Bennett 1996; Mill et al. 2007; Bateman & Langford 1997; Willis et al. 1998; Bateman et al. 1996; Hanley 1989; Hanley & Ruffell 1991; Hanley & Ruffell 1992; Whinteman and Sinclair 1994; Guruluk 2006; Brown 1992; Sutherland and Walsh 1985; Moskowitz and Talberth 1998; Gilbert et al. 1992; Walsh et al. 1984; Clayton and Mendelsohn 1993; Walsh &Loomis 1989; Champ et al. 1997; Loomis & Richardson 2000)* ***Participatory Method*** *(e.g. McDaniels & Roessler 1998)* ***<u>Revealed preference</u>*** ***TCM*** *(e.g. Tobias & Mendelsohn 1991; Loomis 1992; Adger et al. 1995; Kramer et al. 1995; Willis et al. 1998; Zandersen 1997, Chopra 1998; Moskowitz & Talbert 1998; Hadker et al. 1995; Van Beukering et al. 2003; Mano-haran 1996; Manoharan & Dutt 1999; Elasser 1999; Loomis & Ekstrand 1998; Van der Heide et al. 2005; McDaniels & Roessler 1998; Brown 1992; Loomis & Richardson 2000; Yuan & Christensen 1992; Power 1992; Barnhill 1999; Verma, 2008)* ***<u>Production based</u>***		*(e.g Balmford et al., 2003)*	*and Richardson, 2000; Kramer et al., 1995; Murthy & Menkhua, 1994; Dixon & Pagiola 1998; Maharana et a. , 2000; Hanley, Willis, et al., 2002; Garrod and Willins, 1997; Gong, Kontoleon, and Swanson 2003;; Dixon and Sherman, 1990; Adger et al.,1995; Walsh et al. 1984; Kramer & Mercer 1997; Gunawardena et al. 1999; Lockwood et al. 1993;)*

	SERVICES	Wetlands				Forests			
		Direct use	**Indirect use**	**Option use**	**Non-use**	**Direct use**	**Indirect use**	**Option use**	**Non-use**
		1997; Rosenberger and Loomis 2000; Andréassen-Gren & Groth, 1995;)				***Function/Factor Income*** *(e.g. Hodgson & Dixon 1988)* ***Benefits transfer*** ***Production*** ***Benefits transfer*** *(e.g. Walsh & Loomis 1989)*			
20	***Educational*** *(e.g. Opportunities for formal and informal education and training)*	***NA***				***Revealed preference*** ***TCM*** *(e.g. Power 1992)*	***NA***		
21	***Spiritual & artistic inspiration*** *(e.g. source of inspiration; many religions attach spiritual, scared and religious values to aspects of wetland and forest ecosystems)*	***Stated preference*** ***CVM*** *(e.g. Maharana et al., 2000)*				***Revealed preference*** ***TCM & CVM*** *(e.g. Maharana et al., 2000)*			***Stated preference*** ***Contingent Ranking*** *(e.g. Garrod & Willis 1997)* ***CVM / Choice Modelling*** *(e.g. Aaker-lund 2000 by contingent ranking; Mill et al. 2007 by CVM; Kniivila et al. 2002; McDaniels & Roessler 1998; Maharana et al. 2000)* ***Deliberative monetary valuation*** *(e.g. Hanley et al. 2002);*
	Cultural heritage and identity *(e.g. sense of place*	***Stated preference*** ***Choice modelling*** *(e.g. Tuan et al. 2007)*	***NA***				***NA***		

	SERVICES	Wetlands				Forests			
		Direct use	**Indirect use**	**Option use**	**Non-use**	**Direct use**	**Indirect use**	**Option use**	**Non-use**
	and belonging)	***CVM*** *(e.g. Shultz et al. 1998; Tuan et al. 2007)*							
22	***Information for cognitive development***								
Total Economic Value		*(e.g. Kirkland 1988; Thibodeau, Ostro, 1981;Seidl & Morae, 2000; de Groot 1992; Emerton, Kekulandala, 2003;. Costanza et al. 1997)*							
	Combination of Economic Values of Wetlands/Forests	***Benefits Transfer*** *(e.g. Costanza et al. , 1997; Stuip et al. 2002)*	***Benefits Transfer*** *(e.g; Stuip et al. 2002, Seidl and Moraes, 2000; de Groot, 1992)*	***Benefits Transfer*** *(e.g. Costanza et al., 1997; Stuip et al. 2002)*	***Benefits Transfer*** *(e.g. Costanza et al. 1997; Stuip et al. 2002)*	***Benefits Transfer*** *(e.g. Costanza et al. 1997; Stuip et al. 2002, Zandersen et al. 2007, 2009)*	***Benefits Transfer*** *e.g. Costanza et al. , 1997; Stuip et al. 2002)*	***Benefits Transfer*** *e.g. Costanza et al., 1997; Stuip et al. 2002)*	***Benefits Transfer*** *e.g. Costanza et al. 1997; Stuip et al. 2002)*

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Chapter 6

Discounting, ethics, and options for maintaining biodiversity and ecosystem integrity

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Contents

Key messages

- There are no purely *economic* guidelines for choosing a discount rate. Responsibility to future generations is a matter of ethics, best guesses about the well-being of those in future, and preserving life opportunities.

- A variety of discount rates, including zero and negative rates, should be used, depending on the time period involved, the degree of uncertainty, and the scope of project or policy being evaluated.

- In general, a higher discount rate applied to specific cases will lead to the long-term degradation of biodiversity and ecosystems. A 5% discount rate implies that biodiversity loss 50 years from now will be valued at only 1/7 of the same amount of biodiversity loss today.

- But a low discount rate for the entire economy might favor more investment and growth and more environmental destruction.

- In terms of the discounting equation, estimates of how well-off those will be in the future is the key factor as to how much we should leave the future. Policy-makers must decide whether to use income or subjective well-being, or some guess about basic needs.

- A critical factor in discounting is the importance of environmental draw-down (destruction of natural capital) to estimates of g (as GDP growth). Is the current generation living on savings that should be passed to their descendents?

- The rich and poor differ greatly in their direct dependence on biodiversity and ecosystem services and bear different responsibilities for their protection.

1 Introduction

A central issue in the economic analysis of biodiversity and ecosystems is the characterization of the responsibility of the present generation to those who will live in the future. That is, how will our current use of biological resources affect our future life opportunities and those of our descendents? A common approach is to begin with the Brundtland Commission's definition of sustainable development, which emphasizes "meeting present needs without compromising the ability of future generations to meet their own needs" (WCED, 1987). This definition is very general and has widely different interpretations. In economics, sustainability is often interpreted in terms of maintaining human well-being over intergenerational time scales, though some economists attach special importance to conserving stocks of biodiversity, ecosystem services and other forms of natural capital (see Neumayer 2003). As discussed in the TEEB interim report (EC 2008) discounting is a key issue in the economics of biodiversity and ecosystems. How should economists account for the future effects of biodiversity and ecosystem losses using a variety of valuation methods? This leaves open the question of how to integrate traditional cost-benefit analysis with other approaches to understanding and/or measuring environmental values.

For most resource allocation problems economists use a capital investment approach. Resources should be allocated to those investments yielding the highest rate of return, accounting for uncertainty, risk, and the attitude of the investor toward risk. As illustrated in Figure 1, suppose an investor has a choice between letting a valuable tree grow at a rate of 5% per year, or cutting the tree down, selling it, and putting the money in the bank. Which decision is best depends on the rate of interest the bank pays. If the bank pays 6% and the price of timber is constant the investor will earn more money by cutting the tree down and selling it, that is, by converting natural capital into financial capital. This simple example is a metaphor for the conversion of biodiversity and ecosystem services into other forms of capital. The short-comings of this simple approach to valuing biodiversity and ecosystems include

(1) the irreversibility of biodiversity loss,
(2) pure uncertainty as to the effects of such losses,
(3) the difference between private investment decisions and the responsibilities of citizens of particular societies,
(4) the implicit assumption that all forms of capital are in principle substitutable for one another on a Euro-for-Euro basis,
(5) the assumption that reinvestment of natural capital is possible and that future returns on the reinvestment are certain,
(6) the assumption that the change being evaluated is marginal, that is, it will not substantially alter existing economic conditions including relative prices (Hepburn 2006) , and
(7) assuming that the only value of the tree is its potential as timber, thereby ignoring its role in providing ecosystem services. The discount rate can be seen as a reverse interest rate. In the above example, suppose the tree was not growing at all and the rate of interest on money was 6%. By not cutting down the tree and putting the money earned from selling the tree in the bank, the owner would be losing 6% per year. This would be the discount rate on the tree in the world of financial investment.

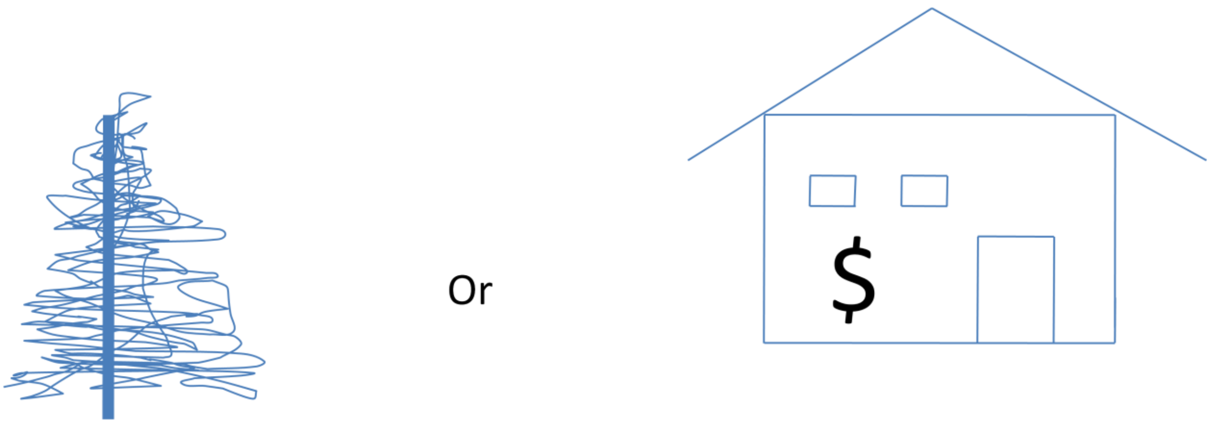

Figure 1. Valuing nature: The tree or money in the bank?

Box 1: Discounting the Amazon

The discussion of the value of the Amazon rainforest illuminates the distinction between economic, cultural, and ecosystem values. The services of the Amazon generate large direct market values, including revenue from ecotourism, fishing, rainforest crops, and pharmaceuticals. Indirect economic benefits include climate regulation, option values for undiscovered rainforest products, and climate change protection from carbon storage. Cultural values not only include the spiritual and life-giving values the Amazon holds for its indigenous people, but also its existence value to the rest of the world's population. Most of those who know something of the unique and beautiful features of Amazon ecosystems feel a loss when they hear of their destruction even if they have never seen them. But the most important values of the Amazon, as discussed in Chapter One, may be its role in providing ecosystem services such as regulating weather in the Western Hemisphere and as the world's largest storehouse of biodiversity.

What role should discounting play in each of these layers of value? As the discussion in Chapter One shows, when calculating the costs and benefits of a development project, such as a dam, discounting will tend to favor short-term economic benefits such as temporary job creation over the costs of losing environmental services that have smaller annual values but that last indefinitely. Cultural and ecosystems values are difficult or impossible to price and thus are usually excluded by traditional cost benefit analysis. Referring to Figure 1, the direct economic services of the Amazon might be represented by the tree. If a section of the Amazon rainforest yields sustainable economic services equivalent to an annual rate of return of 5% and if the forest could be cut down, sold as timber and invested at a rate of return of 6%, then the economically rational thing to do is to cut down the forest, sell it, and put the money in the bank. But this assumes that investments are secure and last indefinitely and that environmental features and economic investments are completely fungible. The tree's role in providing regulating ecosystem services is ignored.

The financial model of resource use is enshrined in the theory of optimal economic growth as described in macroeconomics textbooks (see Blanchard and Fischer 1989, Dasgupta and Heal 1974).

This model assumes that society can and should seek to maximize the weighted sum of present and future economic welfare. The weight attached to future welfare declines at ρ percent per year, reflecting society's impatience, or preference to receive benefits in the short run while deferring costs to the future. In a continuous-time setting with constant population and a single consumption good, this approach employs constrained optimization methods to maximize the social welfare functional:

$$\int_0^\infty U[C(t)]e^{-\rho t}dt \tag{1}$$

subject to the technological, economic, and environmental constraints that force decision-makers to balance short-run and long-run welfare. In this setting, *U* is instantaneous utility and *C* is the flow of consumption goods.[i] This equation characterizes well-being or utility as deriving from the discounted flow of market goods (or pseudo market goods).This characterization of intertemporal choice, particularly in the cases of biodiversity loss and climate change, has been questioned on both theoretical and behavioral grounds (Bromley 1998, DeCanio 2003, Gowdy 2004, Spash 2002). The model ignores the fact that individuals have finite lifespans and assumes that ρ represents both *individuals'* time preference and *social preferences* regarding tradeoffs between the welfare of present and future generations (Burton 1993, Howarth and Norgaard 1992). Although it is restrictive, the discounted utility framework is mathematically tractable. It is perhaps this fact that explains the model's widespread use in applied economics. Most important, the framework is useful for illuminating the economic, ethical, and cultural aspects of valuing the future impacts of public policies.

Beckerman and Hepburn (2007) argue that the practice of utility discounting is justified by the theory of "agent relative ethics." In this perspective, people reasonably attach greater weight to the welfare of themselves and their immediate family than to people who are less proximate to them in space and time. An early contribution by Dasgupta and Heal (1974), however, showed that the discounted utility criterion sometimes generates outcomes that are unsustainable, yielding a moral paradox that has generated a quite substantial literature. This problem arises in economies that depend on an essential, nonrenewable resource, such as oil. In this model, short-run economic growth leads to resource depletion that, in turn, leads to long-run economic decline. This occurs because decision-makers are too impatient to make the investments in substitute technologies needed to offset the costs of resource depletion.

To address this problem, Solow (1974) proposed the so-called "maximum" social welfare function, which allocates resources in a way that achieves a constant level of utility over time. Conversely, equation (1) may be maximized subject to the constraint that utility is constant or increasing over time (Asheim 1988). In this approach, the utility discount rate represents society's altruistic preferences towards future generations, or willingness to undertake voluntary sacrifices so that future generations may enjoy a better way of life. The non-declining utility constraint, in contrast, is based on a perceived moral duty to ensure that present actions do not jeopardize the life opportunities available to posterity (Howarth 1995). This approach can be viewed as rational given a rights-based (or

"Kantian") ethical framework in which moral duties complement preference satisfaction in making rational decisions.

In applied studies, the discounted utility criterion is often embraced as a sufficient basis for optimal resource allocation, a fact that puts special emphasis on the choice of the utility discount rate. The release of the *Stern Review* (Stern 2007) and the ensuing debate among economists as to its merit did much to illuminate the role of discounting the costs and benefits of policies having very long time spans and very broad spatial scales—climate change and biodiversity loss being the prime examples. At first the Stern debate centered primarily on the "proper" discount rate to apply to future costs and benefits of climate change mitigation (Ackerman 2008, Dasgupta 2006, Mendelson 2006-7, Yohe and Tol 2007). As the debate progressed it became clear that there was more to the economics of climate change than choosing the "correct" discount rate. Several prominent environmental economists came to the conclusion that the standard economic model offers an inadequate framework to analyze environmental issues characterized by irreversibilities, pure uncertainty, and very long time horizons (Dasgupta 2008, Weitzman 2009). The "key messages" in the Stern Review's economic analysis of climate change (Chapter 2, p. 25) apply with equal force to the economic analysis of biodiversity. The loss of biodiversity and ecosystems has properties that make it difficult to apply standard welfare analysis including discounting the future:

1. It is a phenomenon having global as well as local consequences.
2. Its impacts are long-term and irreversible.
3. Pure uncertainty is pervasive.
4. Changes are non-marginal and non-linear.
5. Questions of inter- and intra-generational equity are central.

These points have been made about environmental resources for decades by economists working outside the neoclassical paradigm (Boulding 1973, Daly 1977, Georgescu-Roegen 1971). Interestingly, it seems that the policy prescriptions of both those using a more conventional welfare economics approach, and those who call for an alternative, heterodox approach to environmental valuation, are converging. Using either standard or alternative approaches, when the services of nature are taken into account, sustaining human welfare in the future implies aggressive conservation and ecosystem restoration policies in the present.

This chapter is organized as follows. Section 2 discuss the economic approach to intergenerational welfare using the Ramsey discounting equation, section 3 reviews some recent findings from behavioral economics and their relevance to the discounting issue, section 4 examines the issue of ecosystems and biodiversity preservation in the very long run, section 5 discusses the discounting equation in the context of the total value of ecosystems and biodiversity, section 6 examines the macroeconomic implications on biodiversity of a low discount rate, section 7 looks at the issues of discounting and the safe minimum standard, and of ecosystem services and the poor, and section 8 presents conclusions.

2 The Ramsey Discounting Equation and Intergenerational Welfare

In optimal growth theory, it is common to assume that the utility function presented in equation (1) takes the specific form $U(C) = C^{1-\eta}/(1-\eta)$. Here η is a parameter that reflects the curvature of the utility function. Given this assumption, future *monetary* costs and benefits should be discounted at the rate *r* that is defined by the so-called "Ramsey equation":

$$r = \rho + \eta \bullet g \qquad (2)$$

The discount rate r is determined by the rate of pure time preference (ρ), η, and the rate of growth of per capita consumption (g). In intuitive terms, people discount future economic benefits because: (a) they are impatient; and (b) they expect their income and consumption levels to rise so that 1 Euro of future consumption will provide less satisfaction than 1 Euro of consumption today.

This equation ignores uncertainty, thereby streamlining the analysis but reducing the model's plausibility and descriptive power. Accounting for uncertainty leads to a more complex specification in which the discount rate equation includes a third term that reflects the perceived risk of the action under consideration (see Blanchard and Fischer 1989, chapter 6 and Starrtt 1988). As noted above, the rate of pure time preference (ρ) is supposed to reflect both individuals' time preferences and social preferences regarding the value of the well-being of future generations as seen from the perspective of those living today. More realistic models distinguish between these effects, and blending them together can obscure important aspects of both descriptive modeling and prescriptive analysis (Auerbach and Gerlagh and Keyzer 2001, Howarth 1998, Kotlikoff, 1987). A positive value for ρ means that, all other things being equal, the further into the future we go the less the well-being of persons living there is worth to us. The higher the value of ρ the less concerned we are about negative impacts in the future. A large literature exists arguing for a variety of values for pure time preference but it is clear by now that no econometric method is available to determine the value of ρ. Choosing the rate of pure time preference comes down to a question of ethics. Ramsey (1928, 261) asserted that a positive rate of pure time preference was "ethically indefensible and arises merely from a weakness of the imagination." On the other side of the debate, Pearce et al. (2003) took the position that a positive time preference discount rate is an observed fact since people do in fact discount the value of things expected to be received in the future. Nordhaus (1992, 2007) has consistently argued that the market rate of interest constitutes the appropriate discount rate that reveals individuals' time preference.[ii]

Sen's (1961) "isolation paradox" casts doubt on the argument that the social discount rate should be set equal to the market rate of return. According to Sen, private investments may provide spillover benefits that are not captured by individual investors. Providing bequests to one's daughter, for example, serves to increase the welfare of the daughter's spouse and, by extension, his parents. When preferences are interconnected in this way, individuals underinvest. Correcting this market failure would lead to increased investment, lower interest rates, and therefore a lower discount rate in cost-benefit analysis (Howarth and Norgaard, 1993).But even if it is agreed to use a market rate, which market rate should be used? In the United States, a voluminous literature has focused on the fact that, since the late 1920s, safe financial instruments such as bank deposits and short-run government bonds yield average returns of roughly 1%. Corporate stocks, in contrast, yielded average returns of 7% per year with substantial year-to-year volatility. Assets with intermediate risks (such as corporate bonds and long-term government bonds that carry inflation risks) pay intermediate returns. (See Cochrane, 2001, for an authoritative textbook discussion). Choosing a discount rate, then, involves a judgment regarding the perceived riskiness of a given public policy (Starrett 1988). Low discount rates are appropriate for actions that yield safe benefit streams or provide precautionary benefits – i.e., that reduce major threats to future economic welfare.[iii]

Biodiversity loss will affect the entire world's population including those from cultures with very different ideas about obligations to the future. Furthermore, Portney and Weyant (1999, 4) point out that "[t]hose looking for guidance on the choice of discount rate could find justification [in the literature] for a rate at or near zero, as high as 20 percent, and any and all values in between" (quoted in Cole 2008). Frederick, Loewenstein, and O'Donoghue (2004) report empirical estimates of discount rates ranging from -6% to 96,000%. Others argue that discounting from the perspective of an individual at a point in time is not equivalent to a social discount rate reflecting the long term interest of the entire human species. An observed positive market discount rate merely shows that market goods received in the future are worth less as evaluated by an individual living now, not that they are worth less at the point in the future that they are received.

The other important factor in the Ramsey equation determining how much we should care about the future is how well-off those in the future are likely to be. As shown in equation (2), the standard model characterizes the well-being of future generations using two components, the growth rate of per capita income in the future (g) and the elasticity of the marginal utility of consumption (η). The elasticity of marginal utility measures how rapidly the marginal utility of consumption falls as the consumption level increases. It is often assumed that η is equal to 1 (Nordhaus 1994, Stern 2007). In this case, then ηg corresponds to a Bernoulli (logarithmic) utility function, and 1% of today's income has the same value as 1% of income at some point in the future (since g is a percentage change). So if per capita income today is $10,000 and income in the year 2100 is $100,000, $1,000 today has the same value as $10,000 in 2100. Put another way, a $1,000 sacrifice today would be justified only if it added at least $10,000 to the average income of people living in the year 2100 (Quiggin 2008). The higher the value of η, the higher the future payoff must be for a sacrifice today. For example, with ρ near zero and a positive value for g, increasing η from 1 to 2 would double the discount rate.

Several assumptions are buried in the parameter η as it is usually formulated. It is assumed that η is independent of the level of consumption, that it is independent of the growth rate of consumption, and that social well-being can be characterized by per capita consumption. These assumptions are arbitrary and adopted mainly for convenience (Pearce, Atkinson and Mourato 2006).

At least three distinct valuation concepts are present in η (Cole 2008, 18). It contains a measure of risk aversion, a moral judgment about static income inequality among present day individuals, and a moral judgment about dynamic income inequality over time. Weitzman (2009) notes that the values of these components move the discount rate in different directions. On one hand a high value for η (in conjunction with *g*) would seem to take the moral high ground—a given loss in income has a greater negative impact on a poor person than a rich person. But if we assume, as most economic models do, that per capita consumption *g* continues to grow in the future, a higher η means a higher value for η*g* and the less value economists place on income losses for those in the future.

Assuming a near-zero value for ρ and that η = 1 (as in Cline 1992 and Stern 2007) means that the total discount rate (r) is determined by projections of the future growth rate of per capita income, *g*. The growth rate of income is derived from projecting past world economic performance and the researcher's judgment. The values of g in the Stern report and in the most widely used climate change models range between 1.5% and 2.0% (Quiggin 2008, 12). With a high g, discounting the future is justified by the assumption that those living in the future will be better off than those living today (Pearce, Atkinson and Mourato 2006). In the TEEB interim report (EC 2008, p. 30) Martinez-Alier argues that assuming constant growth in *g* leads to the "optimist's paradox." The assumption of continual growth justifies the present use of more resources and more pollution because our descendents will be better off. But such growth would leave future generations with a degraded environment and a lower quality of life.

A major step forward in understanding the economics of sustainability was the realization that maintaining a constant or increasing level of consumption or utility depends on maintaining the stock of capital assets generating that welfare (Arrow et al. 2004, Dasgupta and Mäler 2000, , Hartwick 1977, 1997,Solow 1974). Thus, maintaining a non-declining g means maintaining (1) productive manufactured physical capital, (2) human capital—knowledge, technical know-how, routines, habits and customs—and the institutions supporting it, and (3) natural capital. Using the terminology in Chapter 1, the biodiversity component of natural capital is, in turn, comprised of three different kinds of value to humans:
Economic - The direct inputs from nature to the market economy,
Socio-Cultural - The non-market services necessary for maintaining the biological and psychological needs of the human species, and
Ecological Value - The value to ecosystems such as preserving evolutionary potential through biological diversity and ecosystem integrity.
These layers of biodiversity value are discussed in more detail in section 5.

Under certain technical assumptions, *g* may be interpreted as the growth rate of per capita income adjusted for externalities and other market imperfections. Under these conditions, Dasgupta and Mäler (2000) show that g can be considered as the rate of return on all forms of capital. In a dynamic growth context, *g* is equivalent to the growth rate of total factor productivity (TPF) along a balanced growth path. TFP is the rate of growth of economic output not accounted for by the weighted growth rates of productive inputs. In the three input case used here,

$$TFP = \dot{Q} - a\dot{MK} - b\dot{HK} - c\dot{NK}\text{ , } a+b+c = 1 \text{ and the weights are input cost shares} \qquad (3)$$

For example, in a simple model using manufactured capital (MK), human capital (HK), and natural capital (NK) as inputs, if output grows by 5% per year and the weighted average growth rates of inputs increases by 4%, then TFP would be 1%. Environmental economists have long maintained that estimates of TFP (g in the Ramsey model) do not adequately take into account the draw-down of the stock of natural capital (Ayres and Warr 2006, Dasgupta and Mäler 2000, Repetto et al. 1989). Vouvaki and Xeapapadeas (2008) found that when the environment (they use CO_2 pollution as a proxy for environmental damage) is not considered as a factor of production TFP estimates are biased upward. They argue that failing to internalize the cost of an environmental externality is equivalent to using an unpaid factor of production. After including as natural capital only the external effects of CO_2 pollution from energy use, they found that TFP estimates for 19 of 23 countries switched from positive to negative. The average of TFP estimates for the 23 countries changed from +0.865 to -0.952. This result implies that when the negative effects of economic production on the ability of natural world to provide productive inputs g could well be negative, so future generations would be worse off. Moreover, when environmental degradation effects beyond CO2 accumulation are included the case for a negative g becomes even stronger. This has serious implications for long run economic policies for climate change mitigation and biodiversity loss. Given some reasonable assumptions about pure time preference and the elasticity of consumption, a negative *g* implies that the present generation should consume less in order to invest more in the well-being of future generations.

If we step back from the assumption that the well-being of future generations can be characterized by per capita consumption —by, for example, considering g as representing subjective well-being (Kahneman, Wakker and Sarin 1997)—the case for considering a negative g becomes even stronger. Among the most important findings of the subjective well-being literature are these: (1) traditional economic indicators such as per capita NNP are inadequate measures of welfare; (2) the effect of an income change depends on interpersonal comparisons and relative position; (3) humans have common, identifiable biological and psychological characteristics related to their well-being (Frey and Stutzer 2002, Layard 2005). These observations have direct bearing on the sustainability debate and have the potential to guide intergenerational welfare and policies to protect biological diversity. People receive economic benefits from biodiversity but the psychological and aesthetic benefits cannot be properly captured by market values (Wilson 1994). All these contributions of biodiversity are being rapidly reduced and should be taken into account in estimates of future well-being (*g*). Dasgupta (1995) further identifies self-reinforcing links between environmental degradation, biodiversity loss, poverty, and population growth.

3 Recent Behavioral Literature on Discounting, Risk and Uncertainty

Insights from behavioral economics have greatly broadened understanding of how people compare future costs and benefits. Relevant insights into human behavior include the following:

Loss Aversion - Losses are given higher values than equivalent gains. Figure 2 (from Knetsch 2005) summarizes the considerable evidence that people are loss averse, that is, they evaluate gains and losses from a reference point and place a higher value on a loss than on a gain of an equal amount of that same thing (Kahneman and Tversky 1979). This is confirmed in the widely-reported discrepancy between willingness to pay (WTP) and willingness to accept (WTA) measures of environmental changes (Brown and Gregory 1999). The implications for evaluating biodiversity loss are clear. Even in the context of standard utility theory, the required compensation for biodiversity loss (WTA) is likely to be much greater than the estimated market value of that loss (WTP).

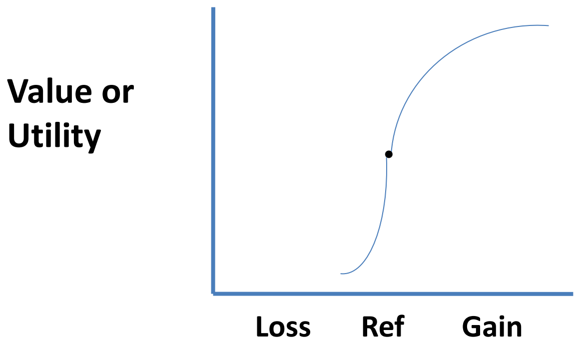

Figure 2 Value of loss or gain from a reference point

Hyperbolic Discounting - Some evidence indicates that people discount the future hyperbolically, that is, as shown in Figure 3, the discount rate declines and then flattens out so that after some time the present value of something no longer significantly declines (Laibson 1997).

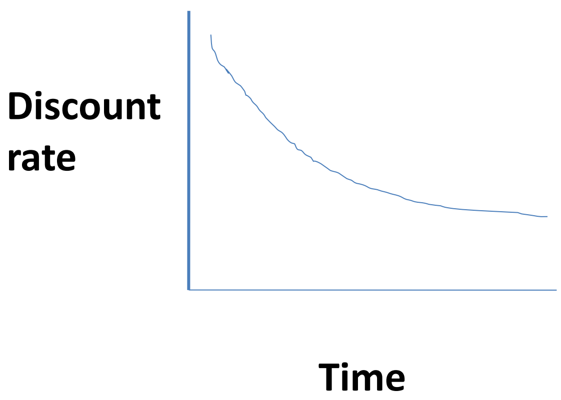

Figure 3 Hyperbolic discounting

The existence of hyperbolic discounting implies that standard economic analysis may seriously underestimate the long-term benefits of biodiversity protection. If people discount hyperbolically, and if we respect stated preferences, straight-line discounting should not be used to place values on distant-future environmental damages such as those caused by biodiversity loss. Hyperbolic discounting has been widely discussed in the theoretical literature and has had some impact on policy recommendations. Cropper and Laibson (1999) recommend using hyperbolic discounting in the case of global warming and Chichilnisky (1996) uses hyperbolic discounting in her model of sustainable development. One of the positive features of welfare economics is that, in theory, it respects individual choice. If individuals choose to place the same value on biodiversity present 50 years from now as they do on biodiversity 100 years from now, then economists should respect that preference. Of course, just because individuals are observed to discount hyperbolically does not mean that a social discount rate should be hyperbolic.
Beltratti, Chichilnisky and Heal (1998) advocate the use of the "green golden rule" for renewable resources, using something near market discount rates in the short run (so that the present is not exploited) and a rate asymptotically approaching zero in the long run (so that the distant future is not exploited). Similarly, Weitzman (2001) advocates what he terms "gamma discounting" using a rate of about 4 percent for the immediate future with a steady decline to near zero in the distant future.

Inconsistent Discounting – Rubinstein (2003) points out that hyperbolic discounting has been accepted by many economists because it can be easily incorporated into the net present value framework of standard economic analysis. He argues that the evidence suggests that the larger problem is inconsistent, not hyperbolic, discounting. People appear to have different discount rates for different kinds of outcomes (Loewenstein 1987). Considerable evidence exists that people are wildly inconsistent even when discounting similar things. Inconsistent discounting suggests some limits to attempts to placing precise numbers on the general tendency of individuals to prefer something now rather than later.

The Equity Premium Puzzle – Mehra and Prescott (1985) showed that the discounted utility model is deeply inconsistent with the observed gap between the low returns available on safe investments and the much higher average returns paid provided by risky assets such as corporate stocks. Mankiw and Zeldes (1991) calculate that the level of risk aversion implied by this rate-of-return spread implies that an investor would have to be indifferent between a bet equally likely to pay \$50,000 or \$100,000 (with an expected value of \$75,000) and a certain payoff of \$51,209.

Explaining the low market return on safe assets requires that both ρ and η must assume values near zero (Kocherlakota, 1996). Yet explaining the high risk premium paid by stocks requires that investors must be highly risk averse, which implies that η must attain a high, positive value to be consistent with the data. Scientifically, this suggests that the discounted utility model is in a deep sense inconsistent with empirical observations. This point undercuts reliance on equation (2) to calculate discount rates. Several approaches have been advanced to address this disparity (see Kocherlakota, 1996; Cochrane, 2001). Some models have extended preferences to distinguish between risk preferences and the elasticity of intertemporal substitution. Others assume that preferences are shaped by habit formation and/or relative consumption effects. A third hypothesis is that investors are loss averse with respect to investment gains and losses (Benartzi and Thaler, 1995). Including this effect in the utility function serves to decouple the social discount rate from the market rate of return (Howarth, 2009). In this case, the public policies should be discounted at a rate that is close to the risk-free rate of return, even for policies that involve significant degrees of uncertainty.

Discounting under Uncertainty – On theoretical grounds, there is reason to believe that greater uncertainty about the future may tend to produce lower certainty-equivalent discount rates (Gollier, 2008). This is because investing in safe assets reduces the risks pertaining to future economic welfare, rendering them attractive to investors even at low rates of return. Newell and Pizer (2003) used random walk and mean-reverting models to compute certainty equivalent discount rates that measure the uncertainty adjusted rate out into the distant future. When applied to climate change scenarios, their results suggested that the present value of mitigation efforts almost doubled. Hepburn et al. (2009) extended this result to estimate autoregressive and regime-switching models of U.S. interest rates and also found that uncertainty-adjusted rates declined more rapidly. Although much of the recent literature on discounting an uncertainty deals with climate change, uncertainty is also pervasive in the case of the welfare effects of biodiversity loss and this suggests using lower discount rates in valuing future losses of biodiversity and ecosystem services.

Discounting and Relative Prices – The discount rate is applied to an aggregate consumption good and an implicit assumption is that the prices of all goods are changing at the same rate. But biodiversity is not a typical consumption good. For at least two reasons—increasing scarcity and limits to its substitutability—calculating the rate of change of the relative *value* of biodiversity will be different from determining the *price* of a typical consumption good (Cameron Hepburn, personal communication). Sterner and Persson (2008, 62) write:

> Briefly, because the rate of growth is uneven across sectors of the economy, the composition of economic output will inevitably change over time. If output of some material goods (e.g. mobile phones) increases, but access to environmental goods and services (e.g. access to clean water, rain-fed agricultural production, or biodiversity) declines, then the relative price of these environmental amenities should rise over time.

This would mean that the estimated damages from biodiversity loss (or the benefits from biodiversity preservation) would rise over time and this might be great enough to offset the effect of the positive discount rate. If this is the case, increasing the amount of biodiversity and ecosystems would be economically justified (Hoel and Sterner 2007).

Risk Aversion and Insurance – A large body of evidence suggests that most people are risk averse (Kahneman and Tversky 1979). This has major implications for evaluating biodiversity and ecosystem losses. As in the case of climate change there is a real, although unknown, possibility that biodiversity loss will have catastrophic effects on human welfare. Paul and Anne Ehrlich (1997) use the "rivet popper" analogy to envision the effects of biodiversity loss. A certain number of rivets can pop out of an airplane body without causing any immediate danger. But once a critical threshold is reached the airplane becomes unstable and crashes. Likewise, ecosystems are able to maintain themselves with a certain range of stress, but after a point they may experience a catastrophic flip from a high biodiversity stable state to another, low diversity stable state. Weitzman (2009) uses the evidence for a small, but significant, possibility of a runaway greenhouse effect to argue for aggressive climate change mitigation policies. Weitzman's reasoning might be applied to ecosystem services considering the possibility of ecosystem collapse once a damage threshold is crossed.

4 Ecosystems and Biodiversity in the Very Long Run

Many economists (for example Spash 2002, chapters 8 and 9) question the appropriateness of discounting as applied to global and far-reaching issues like biodiversity loss. Ultimately, human existence depends on maintaining the web of life within which humans co-evolved with other species and thus the idea of placing a discounted "price" on total biodiversity is absurd. One may object that the ability to adapt to environmental change is one of the most striking characteristics of Homo sapiens (Richerson and Boyd 2005). But the rapidity of current and projected environmental change is unique in human history. Human activity within the past one hundred years or so has drastically altered the course of biological evolution on planet Earth. According to a survey by the International Union for Conservation of Nature, a quarter of mammal species face extinction (Gilbert 2008). Conservative estimates indicate that 12% of birds are threatened, together with over 30% of amphibians and 5% of reptiles. Particularly alarming is the state of the world's oceans. Human-caused threats to ocean biodiversity are summarized by Jackson (2008, 11458):

> Today, the synergistic effects of human impacts are laying the groundwork for a comparatively great Anthropocene mass extinction in the oceans with unknown ecological and evolutionary consequences. Synergistic effects of habitat destruction, overfishing, introduced species, warming, acidification, toxins and mass runoff of nutrients are transforming once complex ecosystems like coral reefs into monotonous level bottoms, transforming clear and productive coastal seas into anoxic dead zones, and transforming complex food webs topped by big animals into simplified, microbially dominated ecosystems with boom and bust cycles of toxic dinoflagellate blooms, jellyfish, and disease.

If we modeled ecosystems according to the Solow-Hartwick approach for economic sustainability (maintaining the capital stock necessary to insure that economic output does not decline) it would certainly be clear that the "ecosystem capital" base for sustaining biodiversity is being rapidly depleted. If the biologists and paleontologists who study the problem are correct, Earth is entering into its sixth mass extinction of complex life on the planet during the past 570 million years or so.

Biodiversity recovery from past mass extinctions took between 5 and 20 million years (Ward 1994, Wilson 1998). Past mass extinctions irreversibly restructured the composition of the earth's biota (Krug, Jablonski and Valentine 2009). Even if the final result of the current mass extinction is a richer, more biologically diverse world, as occurred after past mass extinctions, humans will not be around to see it. Human-induced biodiversity loss will constrain the evolution of humans and other species for as long as humans will have inhabited planet Earth. This prospect raises entirely new kinds of questions about how to value today's impact on future generations. These include:

Functional transparency (Bromley 1989) – In many cases the role of a particular species in an ecosystem is apparent only after it has been removed. The change may be non-linear and irreversible. The effect on local economies may be catastrophic as in the collapse the Northern Cod fishery due to overharvesting. More than 40,000 people in Newfoundland lost their jobs and the cod fishery has still not recovered 15 years after a total moratorium on cod fishing.

Preserving genetic and ecosystem diversity (Gowdy 1997) – Evolutionary potential is the ability of a species or ecosystem to respond to changing conditions in the future. Future conditions are largely unpredictable (the effects of climate change on biodiversity, for example) but in general the greater the diversity of an ecosystem, the more resilient that system is (Tilman and Downing 1994).

Preserving options for future generations (Page 1983, Norton 2005) – The financial model of sustainability treats biodiversity as an input for commodity production. Even if the notion of consumer utility is broadened to include concepts like existence values, the model's frame of reference is still the industrial market economy. The effects of present day biodiversity loss and ecosystem service disruption will last for millennia. The question becomes how do societies decide what to leave for future generations if it is impossible to predict what sorts of economies/values/needs they will have?

5 The Total Value of Ecosystems and Biodiversity and the Discounting Equation

The total value of ecosystems and biodiversity is unknown but logically the ultimate value to humans is infinite because if they are reduced beyond a certain point our species could not exist. Biodiversity value can be seen as layers of a hierarchy moving from market value, to non-market value to humans, to ecosystem value. These various levels of biodiversity value point to the need for a pluralistic and flexible methodology to determine appropriate policies for its use and preservation (Gowdy 1997).

The Economic Value of Ecosystems and Biodiversity

Economic value includes the direct economic contributions of biodiversity including eco-tourism, recreation, and the value of direct biological inputs such as crops, fisheries and forests. These values can be very large. For example, Geist (1994) estimated that the direct economic value of Wyoming's big game animals, from tourism and hunting, exceeded $1 billion or about $1000 for every large animal. Although evidence from contingent valuation, hedonic pricing and other economic valuation tools underscore the importance of biodiversity and ecosystems, these give incomplete, lower-bound estimates of their values (see Nunes and van den Bergh 2001).

The degree to which an economically valuable biological resource should be exploited is driven by the social discount rate, *r* in equation (2) above. The rate is a method for determining (in theory) how to split the stock of natural capital between consumption now and consumption in the future.

The Socio-Cultural Value of Ecosystems and Biodiversity

The biological world contributes to human psychological well-being in ways that can be empirically measured (Kellert 1996, Wilson 1994). But measures of subjective well-being cannot be adequately valued in a traditional social welfare framework (Norton 2005, Orr 2005). Spiritual, cultural, aesthetic, and other contributions of interacting with nature may be included in a more comprehensive conception of utility such as the Bentham/Kahneman notion of utility as well-being.

Considering non-market values of biodiversity helps to answer the question of what should be left for future generations. Is there any reason to think those in the future will not have the same psychological need for interacting with nature? Is there any reason to believe that a walk in a rainforest is worth more to a person living now than to a person living 1, 50, or 100 years from now? The reasonable answer is no. The appropriate discount rate for this part of biodiversity, the pure time preference of biophilia, is $\rho = 0$. Another interesting interpretation of ρ in this context would be consider it as the discount rate if a person were behind a Rawlsian veil of ignorance not knowing where in time she would be placed (Dasgupta 2008).

The Ecological Value of Biodiversity to Ecosystems

Biologically diverse ecosystems seem to be more resilient to environmental shocks than less diverse ones (Tilman and Downing 1994) although the relationship between resilience and biodiversity is complicated (Robinson1992). It is also well established that human activity has degraded terrestrial and marine ecosystems across the planet. Suppose we expand the discounting rule is expanded further to include ecosystem integrity itself? This is reasonable since human existence in the long run depends on preserving our biological context. In the discounting equation (2) suppose "*g*" is considered to be a change in the stock of the Earth's biodiversity and ecosystems. With climate change, continued land clearance and continued exploitation of the world's fisheries, *g* is likely to be negative for decades to come. Let η be the value of a marginal change in the state of an ecosystem. The value of η is likely to be larger the more degraded an ecosystem is (more susceptible to changes in the state of the environment). So the term ηg applied to ecosystems is likely to be negative, large, and increasing in the future. This implies making large sacrifices today to improve ecosystems in the future. Today's generation has prospered by spending much of the natural capital it inherited. Ethically, they owe it to future generations to rebuild that inheritance.

6 Does a low Discount Rate Promote Conservation?

The discount rate is also relevant to investment and economic performance at the macroeconomic level which in turn affects biodiversity and ecosystems. At the macroeconomic level there is no unambiguous relationship between the rate of interest (the mirror image of the discount rate) and the extent of biodiversity conservation. A low rate of interest can be associated with a high degree of biodiversity loss and so can a high rate of interest. This follows if the level of investment in human-made capital is regarded as a major factor leading to ecosystem disruption and biodiversity (Tisdell, 2005, p. 250). That the accumulation of manufactured capital is a major factor resulting in species loss has been pointed out for a long time (Harting 1880, p. 209, Swanson 1994, Tisdell 1982, p.378, 1991). Manufactured capital is a produced input using "land" (the direct use of biodiversity and ecosystems, and their indirect destruction) and labor.

For simplicity, assume that the real rate of interest depends only on the demand for loanable funds for investment and on the supply of these funds as a result of savings. Assume further that these demand and supply curves have normal slopes. First, it can be observed that in this case, an increase in the rate of interest can come about either because the demand for loanable funds rises (due to an increase in the marginal efficiency of capital), other things kept constant, or due to fall in the willingness to save, other things unchanged. These two situations are illustrated in Figures 4and 5 respectively. In the case shown in Figure 4, the demand for loanable funds rises from D_1D_1 to D_2D_2 and the supply curve of these funds remains unaltered as shown by S_1S_1. The equilibrium in the loanable funds market changes from E_1 to E_2 and the rate of interest rises from r_1 to r_2. The amount of funds invested goes up from X_1 to X_2. This result is unfavorable to biodiversity conservation because it results in more capital accumulation and conversion of natural resources into man-made capital. On the other hand, in the case illustrated by Figure 4, a rise in the rate of interest is associated with a reduction in the level of investment and therefore is favorable to biodiversity conservation.

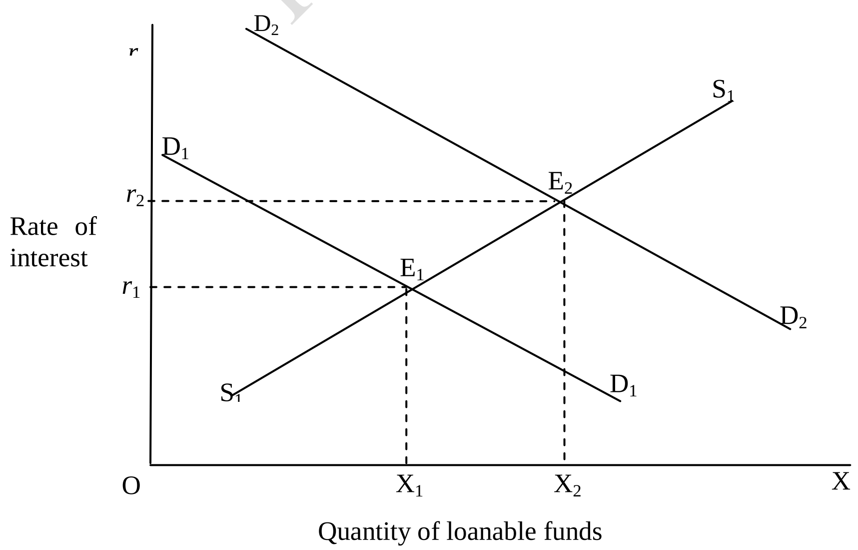

Figure 4 A case in which a rising rate of interest is associated with a rise in the level of investment in man-made capital; a consequence likely to have an adverse impact on biodiversity conservation.

In this case, the demand curve for loanable funds, D_1D_1 is stationary but the willingness to supply loanable funds declines, as shown by the supply line being initially S_1S_1 and subsequently S_2S_2. Market equilibrium alters from E_1 to E_2 and the rate of interest rises from r_1 to r_2. However, in this case, the level of investment falls from X_1 to X_0, and the result is favorable to biodiversity conservation.

Converse results also apply. If the demand curve in the case illustrated in Figure 4 shifts downwards rather than upwards, the rate of interest falls but investment does likewise. On the other hand, if the supply curve of loanable funds moves downwards in the case illustrated by Figure 5, the interest rate falls but the level of investment rises. The fall in the interest rate in the former case is favorable to biodiversity conservation but not in the latter one.

Other examples could also be given. The ones above, however, are sufficient to show that at the macro-level, changes in the rate of interest can be associated (depending on the circumstances) with an increase or decrease in the level of investment in man-made capital. If investment in human-made capital is seen as the main threat to biodiversity and ecosystem conservation (a reasonable proposition) then it can be concluded that the level of the real rate of interest is not closely connected to the degree of biodiversity conservation. This suggests that, at the macroeconomic level, the focus of concern ought to be on variations in the level of human-made capital rather than on the rate of interest as a major influence on biodiversity conservation.

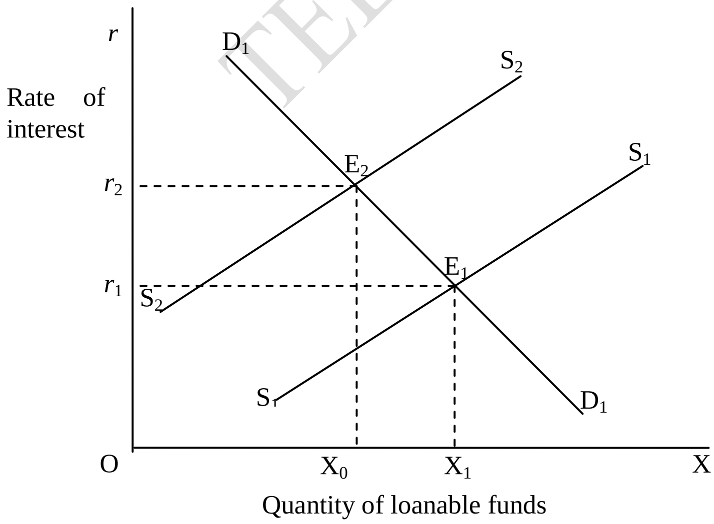

Figure 5 A case in which a rise in the rate of interest is associated with a decline in the level of investment in man-made capital. This case is likely to be favorable to biodiversity conservation.

This point, however, provides only a limited insight into the determinants of capital accumulation. For instance, savings and investment levels tend to rise as aggregate income increases. Investment is usually the basis for further capital accumulation because of its impact on economic growth – rising incomes result in greater levels of saving and investment. Massive increase in capital accumulation since the Industrial Revolution has had extremely adverse consequences for the conservation of biodiversity.

Keynes (1936, Chapters 16 and 24) thought it possible that capital could accumulate in modern times to such an extent that the marginal efficiency of capital would become zero. But of course, he only had in mind manufactured capital. This would result in Keynes' view in the rate of interest being zero or close to it. Yet it can be hypothesized that in order to reach this stationary state would require a tremendous conversion of natural resources into human-made capital resulting in great biodiversity loss. Consequently, a zero rate of interest can be associated in this instance with major loss of biodiversity. This observation reinforces the position of the TEEB interim report (EC 2009) that a variety of discount rates are needed depending on the scale (economy-wide, local community or individual), the timeframe (immediate or distant future), and income group being considered (rich or poor).

7 Discounting and Safe Minimum Standards

Resource economics has a long tradition in of applying a higher discount rate to the benefits of development and a lower rate to the environmental costs of that development. Fisher and Krutilla (1985) suggest a formula for estimating the net present value for a development project that reduces to:

$$NPV(D) = -1 + D/(r+k) - P(r-h) \tag{4}$$

Where D is the value of development and P is the value of preservation. In this setup, a factor k is added to the discount rate applied to development benefits to reflect the depreciation of development benefits over time. In a similar vein, a factor h is subtracted from the rate of discount applied to the benefits of preservation. Here h is supposed to represent growth in the value of environmental services over time based on increased material prosperity that augments willingness to pay for scarce nonmarket goods. No hard and fast rules can be applied to determine exactly how much these discount rates should be adjusted.

This status quo bias mentioned above lends support to the notion of a safe minimum standard (SMS) and the precautionary principle. The SMS approach (Bishop 1978) explicitly recognizes that irreversible environmental damage should be avoided unless the social costs of doing so are "unacceptably high." The concept is necessarily vague because it does not rely on a single money metric. It recognizes that a discount premium should be applied to environmental losses, economic gains should be discounted more heavily, a great amount of uncertainty is involved in judging the effects of environmental losses, and there are limits to substituting manufactured goods for environmental resources.

Rights-based or deontological values are widely held, as indicated by numerous valuation surveys (Lockwood 1998, Spash 1997, Stevens et al. 1991). A rights-based approach may be especially appropriate for policies affecting future generations (Howarth 2007, Page 1983). Do future generations have a right to clean air, clean water, and an interesting and varied environment? There is no reason to think that future generations would be any more willing than the current one to have something taken away from them forever (especially things like a stable climate and biological species) unless they are compensated by something "of equal value." A rights-based approach to sustainability moves away from the welfare notions of tradeoffs and fungibility toward the two interrelated concerns of uniqueness and irreversibility. As Bromley (1998, 238) writes: "Regard for the future through social bequests shifts the analytical problem to a discussion about deciding what, rather than how much, to leave for those who will follow." The question of what to leave also moves away from marginal analysis, and concern only about relative amounts of resources, toward looking at discontinuous changes and the basic biological requirements of the human species in evolutionary context.

It is often argued that only the wealthy have the luxury to be concerned about environmental quality. This is the so-called "post-materialist" thesis of Inglehart (1990), Krutilla (1967) and others. Martinez-Alier (1995) points out two flaws in this thesis. First, continued material growth implies environmental degradation, and second, the world's poorest receive a large percentage of their livelihoods directly from ecosystems (firewood, water, fish and game). The "GDP of the poor" (EC 2008) is undervalued because so much of it depends on un-priced inputs from nature. Yet, in general it is the rich who decide whether or not to preserve biodiversity and ecosystems based on market rates of interest and investment opportunities. On one hand, the poor have a larger stake in preserving flows of ecosystem services, but on the other hand a large percentage of the world's poor are in such a desperate position that they have very high discount rates. Cultural differences are also critical, not only between the rich and poor but also among the diverse cultures of the world's poorest. Traditional cultures in general have a reverence for the natural world but specific practices and cultural attitudes toward nature vary considerably. Likewise large variations exist in the environmental attitudes of the wealthy (Bandara and Tisdell 2004).

Table 1 is a broad attempt to translate these general observations into discount rates. Here, "rich" generally means the middle and upper middle class population of North America, Japan, and Europe. A sizable portion of the world's wealthiest may not fit this category. "Poor" generally means the 1 billion or so of the world's population living on less than $1 a day (World Bank estimate). This group is also heterogeneous. The bottom line is that characterizing responsibilities to future generations by a "discount rate" does not do justice to the nuances of human cultures, the heterogeneous nature of the many contributors to well-being, or the pure uncertainty as to the future of *Homo sapiens* on planet earth.

Table 1 General Observation about Life Opportunities and Discounting

	Ecosystem Services	Manufactured Capital
Poor	ρ = low in the case of traditional cultures	ρ = low, maintain capital stock for future generations
	ρ = positive for cultures under severe stress	ρ = positive – cultures under stress may be unable to invest in capital stock maintenance
	η = likely to be very high any increment to consumption will add to well-being	η = very high
	g = for the very poor, likely to be negative (future generations will be worse off because of deteriorating environmental conditions, but this cannot be taken into account in economic decisions). Consider its value to be 1 so that ng is large.	
Rich	ρ = 0 (ethical responsibility to future generations	ρ = 0 or (-) maintain or increase natural capital stock for the future
	η = positive, higher incomes imply higher income elasticities for environmental goods	η = 0 or negative. More capital means more consumption, but not an increase in well-being
	g = negative for the very rich to compensate for past natural capital destruction, also even if future generations have more material wealth,	
	does this mean they need less biodiversity?	

8 Summary of the Major Challenges to Discounting Biodiversity and Ecosystem Losses

Even a few years ago economists were quite confident about the ability of the standard economic model to capture the future values of environmental features. But recent debates among economists over two of the most pressing issues of our time, biodiversity loss and climate change, have made it clear that no purely *economic* guidelines are available for valuing essential and irreplaceable features of the natural world. Responsibility to future generations is a matter of inter- and intra-generational ethics, best guesses about the well-being of those who will live in the future, and preserving life opportunities for humans and the rest of the living world. Economics can offer valuable insights, as the discussion surrounding the Ramsey equation has shown, but ultimately economic value represents only a small portion of the total value of biodiversity and ecosystems. The practice of discounting applies first and foremost to an individual deciding how to allocate scarce resources at a particular point in time. In general, an individual would prefer to have something “now” rather than in the future, though with some exceptions (the value of anticipation, for example). This is the main argument for a positive discount rate. But, again in general, a higher discount rate will lead to the long-term degradation of biodiversity and ecosystems. For example, a 5% discount rate implies that biodiversity loss 50 years from now will be valued at only 1/7 of the same amount of biodiversity loss today. This leads to the following observations:

1. There is a fundamental difference between the individual-at-a-point-in-time discount rate and the social discount rate. The Ramsey equation ($r = \rho + \eta \cdot g$ as discussed above) can help to illuminate this difference. Ethical responsibility to future generations is captured, in part, by the term ρ, the rate of pure time preference. Although there is still considerable disagreement among economists, a strong case can be made that ρ should be near zero, indicating that there is no reason to place a lower value on the well-being of a person who happens to be born later in time than another person.

2. In terms of the discounting equation, estimates of how well-off those in the future will be (*g*) is the key factor as to how much we should leave the future. Should we use income or subjective well-being, or some guess about basic needs? *g* should encompass everything that gives people utility including intangible benefits of nature (Dasgupta and Mäler 2000). In practice, however, per capita consumption is usually used as a proxy for well-being (Stern 2007).

3. A critical factor in discounting is the importance of environmental draw-down (destruction of natural capital) to estimates of the future growth rate of per capita consumption, *g*. Some evidence indicates that the current generation has prospered by drawing down savings (natural capital) that should have been passed to our descendents. A case can be made that estimates of *g* (and of the discount rate) should be negative.

4. In contrast to the recommendations of conventional economists, a variety of discount rates, including zero and negative rates, should be used depending on the time period involved, the

degree of uncertainty, ethical responsibilities to the world's poorest, and the scope of project or policy being evaluated.

5. A low discount rate for the entire economy might favor more investment and growth and more environmental destruction. Macroeconomic consequences of a particular discount rate should be considered separately from microeconomic ones.

6. The rich and poor differ greatly in their direct dependence on biodiversity and the services of ecosystems. The world's poorest, probably numbering in the billions, live to a large extent directly on ecosystem services and biodiversity (Gundimeda and Sukdev 2008). These people are suffering disproportionally from the loss of ecosystems and biodiversity.

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[i] Equation (1) will take different forms depending on whether we consider discrete or continuous time, whether or not we allow population growth, or whether we consider per capita consumption or total consumption. See the discussion in Arrow, Dasgupta and Mäler 2003.

[ii] Hepburn (2006) suggests that the recent literature on “optimal paternalism” calls for correcting “internalities”, that is, behavior damaging to the individual. A lower discount rate might be imposed by a paternalistic government to insure the optimal preservation of biological diversity.

[iii] On the other hand Lind (1982) argues that the risk and insurance aspects of an investment should not be accounted for by adjusting the social rate of time preference but rather by adjusting the estimates of costs and benefits.

The Economics of Ecosystems and Biodiversity

The Ecological and Economic Foundation

Chapter 7

Key Messages and Linkages with National and Local Policies

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Contents

Synthesis

Synthesis

One of the major objectives of this TEEB book is to assess current approaches for using ecological sciences and economics for informed choices and decision making. On the one hand TEEB intends to better inform conventional economic policy about its impacts on ecosystem health and biodiversity, on the other it suggests ways to mainstream the valuation of ecosystem services into national and local planning and policies as well as business assessments of their economic impacts and dependencies on biodiversity. The previous six chapters assess the state of art of our scientific understanding underlying economic analysis of ecosystem services and biodiversity. Chapter 1 in this volume identifies several challenges in integrating the disciplines of ecology and economics and organizes the complexities of the problem resulting from the differences in methodological frameworks in relation to variation in temporal and spatial scales. For example, the relevant time horizon for a cost–benefit analysis of afforestation or ecosystem restoration project is 10–20 years while the changes in biodiversity could occur on a time scale ranging from short – a fraction of a second (molecular level) to long – millions of years (biome level). Chapter 2 highlights that ecosystems typically produce multiple services that interact in complex ways. A resilient ecosystem maintains a flow of ecosystem services on a continuous basis, but ecosystems may or may not be resilient to anthropogenic disturbances. Chapter 3 suggests that biodiversity and ecosystems are amenable to economic analysis only if they can be quantified and several approaches for developing indicators are discussed. Chapters 4 and 5 together highlight the socio-cultural embeddedness of ecosystem service and biodiversity valuation, as well as its constraints and limitations. Chapter 4 outlines the role and importance of valuation as a human institution and places in context the role of *economic* valuation. Chapter 5 explains how, for a chosen set of ecosystem indicators and within a structured economic valuation framework, there could be reliable approaches for the economic valuation of ecosystem services.. Chapter 5 also highlights commonly practised methods of ecosystem service valuation, and discusses their constraints and limitations. Chapter 6 assesses the basis of choosing appropriate discount rates to be applied to a project with impacts on ecosystems and biodiversity.

In this chapter we attempt to summarize the key lessons learned from the assessments done in Chapters 1–6. The summary can be categorized under the following headings:

1 Framing of issues for economics of ecosystems and biodiversity

The study of the economics of ecosystem services and biodiversity strongly emphasizes the joint effort of ecology and economics. There is a growing need for collaboration between ecologists and economists to have a coherent perspective on the trade-offs reflected in individual and societal choice (Polasky and Segerson, 2009). Even if the outlooks of the two disciplines differ, the scale of the problem requires that solutions can only emerge in making the methodology more porous and fluid in order to embrace each other's take on the problem. We find that:

1. Linking biophysical aspects of ecosystems with human benefits through the notion of ecosystem services is essential to assess the trade-offs involved in the loss of ecosystems and biodiversity in a clear and consistent manner.

2. Economic assessment should be spatially and temporally explicit at scales meaningful for policy formation or interventions, inherently acknowledging that both ecological functioning and economic values are contextual, anthropocentric, individual-based and time specific.

3. Economic assessment should first aim to determine the service delivery in biophysical terms, to provide ecological underpinning to the economic valuation or measurement with alternative metrics.

4. Clearly distinguishing between functions, services and benefits is important to make ecosystem assessments more accessible to economic valuation, although no consensus has yet been reached on the classification.

5. Ecosystem assessments should be set within the context of contrasting scenarios – recognizing that both the values of ecosystem services and the costs of actions can be best measured as a function of changes between alternative options.

6. In assessing trade-offs between alternative uses of ecosystems, the total bundle of ecosystem services provided by different conversion and management states should be included.

7. Any valuation study should be fully aware of the 'cost' side of the equation, as focus on benefits only ignores important societal costs such as missed opportunities of alternative uses; this also allows for a more extensive range of societal values to be considered.

8. Economic assessments should integrate an analysis of risks and uncertainties, acknowledging the limitations of knowledge on the impacts of human actions on ecosystems and their services and on their importance to human well-being.

9. In order to improve incentive structures and institutions, the different stakeholders – that is, the beneficiaries of ecosystem services, those who are providing the services, those involved in or affected by the use, and the actors involved at different levels of decision making – should be clearly identified, and decision making processes need to be transparent.

10. Efforts aimed at changing behaviour of individuals and society towards their impact on ecosystems and biodiversity must take into account that ecosystems have always been dynamic, both internally and in response to changing environments.

11. The importance of using scenarios in ecosystem service assessments is beginning to be realized as early assessments presented a static picture in a rapidly changing world. The necessity of providing counter-factual evidence is now being demanded of conservation research (Ferraro and Pattanayak, 2006) and should become the norm in ecosystem service research as well.

12. The generation of scenarios is particularly important for monetary valuation, since scenarios enable analysis of changes in service delivery which are required to obtain marginal values. Making an analysis in incremental terms avoids (or at least reduces) the methodological difficulties which arise, depending on the relative magnitude of changes, especially when attempting to estimate total values given the non-constancy of marginal values associated with the complete loss of an ecosystem service.

13. In the TEEB context, comparing the outputs under several scenarios will inform decision makers of the welfare gains and losses of alternative possible futures and different associated policy packages.

14. *Indirect drivers* of ecosystem change include demographic shifts, technology innovations, economic development, legal and institutional frameworks, including policy instruments, the steady loss of traditional knowledge and cultural diversity and many other factors that influence our collective decisions.

15. All ecosystems are shaped by people, directly or indirectly and all people, rich or poor, rural or urban, depend directly or indirectly on the capacity of ecosystems to generate essential ecosystem services. In this sense, people and ecosystems are interdependent social-ecological systems. It is not surprising that around 1.2 billion poor people are located in fragile and vulnerable ecosystems but it is not primarily the actions of the poor that affect those fragile ecosystems but rather the actions of 'rich' people who interfere with them and put in question the dependence of the poor (Barbier, 2008).

2 Linkages of ecosystem, ecosystem services and biodiversity

A clear understanding of the links between ecosystems and ecosystem services, and how variation in biodiversity affects ecosystem dynamics is needed to map the temporal and spatial flow of services. This will not only help in making robust valuations but would avoid double counting and provide necessary caveats while up-scaling the value from local to national and regional scale. We find that:

1. Variation in biological diversity relates to the function of ecosystems in at least three ways: (i) increase in diversity often leads to an increase in productivity due to complementary traits among species for resource use, and productivity itself underpins many ecosystem services; (ii) increased diversity leads to an increase in response diversity (range of traits related to how species within the same functional group respond to environmental drivers) resulting in less

variability in functioning over time as environment changes; (iii) idiosyncratic effects due to keystone species properties and unique trait-combinations which may result in a disproportional effect of losing one particular species compared to the effect of losing individual species at random.

2. Ecosystems produce multiple services and these interact in complex ways, different services being interlinked, both negatively and positively. Delivery of many services will therefore vary in a correlated manner, but when an ecosystem is managed principally for the delivery of a single service (e.g. food production) other services are nearly always affected negatively.

3. Ecosystems vary in their ability to buffer and adapt to both natural and anthropogenic changes as well as recover after changes (i.e. resilience). When subjected to severe change, ecosystems may cross thresholds and move into different and often less desirable ecological states or trajectories. A major challenge is how to design ecosystem management in ways that maintain resilience and avoid the passing of undesirable thresholds.

4. There is clear evidence for a central role of biodiversity in the delivery of some – but not all – services, viewed individually. However, ecosystems need to be managed to deliver multiple services to sustain human well-being and also managed at the level of landscapes and seascapes in ways that avoid the passing of dangerous tipping-points. We can state with high certainty that maintaining functioning ecosystems capable of delivering multiple services requires a general approach to sustaining biodiversity, in the long term also when a single service is the focus.

5. When predicting the impact of biodiversity change on variability in the supply of ecosystem services, we need to measure the impact of biodiversity conservation over a range of environmental conditions. In the same way, we need to be able to identify the effect of biodiversity change on the capacity of social-ecological systems to absorb anthropogenic and environmental stresses and shocks without loss of value (Scheffer et al, 2001; Kinzig et al, 2006).

6. To understand and enhance the resilience of such complex, coupled systems, we need robust models of the linkages between biodiversity and ecosystem services, and between biodiversity change and human well-being (Perrings, 2007; Perrings et al, 2009).

3 Choice of indicators and value-articulating institutions

Economic analysis, especially the evaluation of changes in ecosystem services due to marginal change in intervention and policies, requires careful choice of indicators and measures. The social and cultural contexts in which value articulating institutions exist and reveal value must be known to those assigning economic values. Establishing these prerequisites would also provide greater credibility to the estimates that follow. We find that:

1. A lack of relevant information at different scales has hampered the ability to assess the economic consequences of the loss of ecosystems and biodiversity.

2. Most of the current measures and indicators of biodiversity and ecosystems were developed for purposes other than the economic assessment. They are therefore unable to show clear relationships between components of biodiversity and the services or benefits they provide to people.

3. A reliance on these existing measures will in all likelihood capture the value of only a few species and ecosystems relevant to food and fibre production, and will miss out the role of biodiversity and ecosystems in supporting the full range of benefits, as well as their resilience into the future.

4. A set of indicators is needed that is not only relevant and able to convey the message of the consequences of biodiversity loss, but must also be based on accepted methods that reflect the aspects of biodiversity involved and the service that is of interest, capture the often non-linear and multi-scale relationships between ecosystems and the benefits that they provide, and be convertible into economic terms.

5. While it is possible to obtain preliminary estimates of the consequences of biodiversity and

ecosystem loss using existing data and measures, these must be complemented with active research and development into the measurement of biodiversity and ecosystem change, their links to benefit flows and the value of these flows so as to realize the full value of biodiversity and ecosystem management

6. The flow of ecosystem services from point of production to point of use is influenced by both biophysical (e.g. currents, migration) and anthropogenic (e.g. trade, access) processes which influence the scale of service flow from locally produced and used services (e.g. soil production) to globally distributed benefits (e.g. carbon sequestration for climate regulation).

7. In order to make a comprehensive and compelling economic case for the conservation of ecosystems and biodiversity it is essential to be able to understand, quantify and map the benefits received from ecosystems and biodiversity, and assign values to those benefits.

8. Biophysical measurements are important since biodiversity underpins the delivery of many ecosystem services and thus forms the underlying basis of value.

9. Valuation, including economic valuation, functions as a system of cultural projection which imposes a way of thinking and a form of relationship with the environment and reflects particular perceived realities, worldviews, mind sets and belief systems. However, it can also serve as a tool for self-reflection and feedback mechanism which helps people rethink their relations to the natural environment and increase knowledge about the consequences of consumption, choices and behaviour.

10. Due to multidimensional and socio-cultural embeddedness of value any exercise of valuation is relative to a given individual or group of people. In a multicultural and democratic context of biodiversity valuation, this makes the question of choosing a value-articulating institution more important than that of finding a correct value.

11. Economic valuation influences the notion of ownership and property applied to biodiversity and over the long term may change human relationship to the environment in significant ways.

12. Intrinsic values are culturally embedded. They can be taken into account by choosing the appropriate institutions which allow their articulation in addition to utilitarian values.

13. Valuation processes can be seen as a form of regulatory adaptation by serving as a mechanism to provide feedback in a an economic system where production and consumption, trade and exchange are so distant from the underlying ecosystem and so complex in their commercial structure that they may undermine perceptions of the impacts of human habits and behaviours on the natural environment.

14. Value change along the commodity chain has implications for the distribution of benefits, affects the level of incentives for conservation and represents an important methodological challenge for economic valuation.

15. Economic valuation may contribute to address our inability, reluctance or ideological intolerance to adjust institutions (also those which are value articulating) to our knowledge of ecosystems, biodiversity and the human being.

16. Economic valuation is a complex, spatial and institutional cross-scale problem. Many efforts focusing on particular parts of ecosystems or species, while effective at one level, lack the scope to control the pressure of commodity markets for land resources surrounding them. As such, and depending on their biophysical context, they may be limited to capturing the linkages and vertical interplay created by a growing functional interdependency of resource-use systems nested within larger ecosystems.

4 Economic value, valuation methods, non-linear changes, resilience and uncertainty

How to value ecosystem services, what are the tools available, and what are the assumptions necessary to arrive at credible and transparent estimates and finally come out with recommended methods to apply in a situation of non-linear changes remain the main mandate of TEEB. The literature on valuation is full of good suggestions (Freeman, 2003; Heal et al, 2005; Barbier, 2007; Hanley and Barbier, 2009; Barbier, 2009; Atkinson, 2010, Bateman et al, 2010). Yet valuation techniques face important challenges especially regarding uncertainty, irreversibility and resilience. They are typically found while valuing the regulating services of ecosystems (Kumar and Wood, 2010). Our assessment suggests that:

1. Estimating the value of the various services and benefits that ecosystems and biodiversity generate may be achieved with a variety of valuation approaches. Applying a combination of approaches may overcome disadvantages of relying solely on an individual method.

2. Valuation techniques in general and stated preference methods specifically are affected by uncertainty stemming from gaps in knowledge about ecosystem dynamics, human preferences and technical issues in the valuation process. There is a need to include uncertainty issues in valuation studies. However, when uncertainty is compounded by ignorance about ecosystem functioning or when there is even a small possibility of disastrous damages, such as complete ecological collapse of ecosystems, current valuation techniques used to estimate values to feed into extended cost–benefit analyses are insufficient.

3. Valuation results will be heavily dependent on social, cultural and economic contexts, the boundaries of which may not overlap with the delineation of the relevant ecological system. It is likely that better valuation can be achieved by identifying and involving relevant stakeholders.

4. Implementation of valuation approaches that may be suitable for some developed regions may be inadequate in developing countries, thus not being immediately transferable. Hence, standard valuations approaches need to be carefully adapted to account for particular challenges that arise in developing countries.

5. While benefits transfer methods may seem a practical, swift and cheaper way to get an estimate of the value of local ecosystems, particularly when the aim is to assess a large number of diverse ecosystems, due care should be exercised in their use especially when key features of ‘sites’, such as ecosystem dynamics, socio-economic and cultural contexts, largely differ one from another.

6. Benefit transfer methods can be divided into four categories in increasing order of complexity: (i) unit BT; (ii) adjusted unit BT; (iii) value function transfer; and (iv) meta-analytic function transfer. BT using any of these methods may result in estimates that differ from actual values, so-called transfer errors. The acceptable level of transfer error for decision making is context-specific, but if a highly precise value estimate is required it is recommended to commission a primary valuation study.

7. Economic valuation can provide useful information about changes to welfare that will result from ecosystem management actions, especially with regard to localized impacts that are fairly well known and far from ecological thresholds.

8. Valuation practitioners should acknowledge that valuation techniques face limitations that are as yet unresolved. They should present their results as such, and decision makers should interpret and use valuation data accordingly.

9. The limitations of monetary valuation are especially important as ecosystems approach critical thresholds and ecosystem change is irreversible, or reversible only at extreme cost. In this case and until more understanding of ecological dynamics and techniques to estimate the insurance value of biodiversity or the value of ecosystem resilience become available, under conditions of high uncertainty and existence of ecological thresholds, policy should be guided by the 'safe-minimum-standard' and 'precautionary approach' principles.

10. There are three sources of uncertainty pervading valuation of ecosystem services and biodiversity that have been taken into account: (i) uncertainty regarding the delivery or supply of ecosystem services and biodiversity; (ii) preference uncertainty; and (iii) technical uncertainty in the application of valuation methods. Some promising approaches are being developed to try to account for such types of uncertainty but generally valuation applications disregard the uncertainty factor.

11. The uncertainty regarding the delivery of ecosystem services makes stated preference methods complex. Stated preference methods have generally resorted to measuring respondents'

risk perceptions. Other valuation approaches based on expected damage functions are based on risk analysis instead.

12. Preference uncertainty is inversely related to the level of knowledge and experience with the ecosystem service to be valued. This source of uncertainty has been relatively better acknowledged in stated preference approaches, for instance by requesting respondents to report a range of values rather than a specific value for the change in the provision of an ecosystem service.

13. Technical uncertainty pervades valuation studies especially with regard to the credibility of the estimates of non-use values through stated preference methods and the non conclusive issue of the large disparity between WTP and WTA value estimates. It has been suggested that combining valuation models and a preference calibration approach may be the way forward to minimize technical uncertainty.

14. The value of the resilience of an ecosystem is related to the expected benefits and costs that occur when the ecosystem shifts to another regime. An analogy can be drawn between the valuation of ecosystem resilience and the valuation of a portfolio of assets in that the value of the asset mix – the ecosystem and its biodiversity – depends on the probability that a shift occurs as well as on the benefits and costs when it does.

15. Current knowledge about biodiversity and ecosystem dynamics at this point is insufficient to implement such portfolio assessment, and monetary analysis will be misleading when ecosystems are near critical thresholds. At the policy level, it is better to address this uncertainty and ignorance by employing a safe minimum standard approach and the precautionary principle.

16. Despite many limitations, valuation exercises can still provide information that is an indispensable component of environmental policy in general. But policy makers should interpret and utilize the valuable information provided by these techniques while acknowledging the limitations of this information.

17. It is likely that new techniques and combinations of different methodological approaches (e.g. monetary, deliberative and multicriteria methods) will be needed in order to properly face future challenges and provide more accurate values that would benefit decision making processes.

18. A closer collaboration between ecologists and economists may then contribute to develop valuation techniques that are better suited to dealing with the complex relationship between ecosystems and the services they provide to the local and global economies.

19. Future valuation practitioners of biodiversity and ecosystem services should make explicit the procedures and methods used in their studies as well as openly acknowledge any obstacles that they may have encountered.

5 Discounting as an ethical choice

Intertemporal distribution of costs and benefits poses a challenge for the decision makers in justifying resource allocation for a project and policies especially when they have competing demand for the resource available. The challenge becomes more severe if the project entails the impact on ecosystems in the long run. If the impacts of project through its costs and benefits accrue to poor and the rich disproportionately, it further complicates the analysis.

We find that an appropriate rate of discount can guide better choice of strategies and response policies for ecosystem management. We suggest:

1. There is a fundamental difference between the individual-at-a-point-in-time discount rate and the social discount rate.

2. In terms of the discounting equation, estimates of how well-off those in the future will be are the key factor as to how much we should leave the future.

3. A critical factor in discounting is the importance of environmental draw-down (destruction of natural capital) to estimates of the future growth rate of per capita consumption

4. In contrast to the recommendations of conventional economists, a variety of discount rates, including zero and negative rates, should be used depending on the time period involved, the degree of uncertainty, ethical responsibilities to the world's poorest, ethical responsibilities towards future generations, and the scope of project or policy being evaluated.

5. A low discount rate for the entire economy might favour more investment and growth and more environmental destruction. Macroeconomic consequences of a particular discount rate should be considered separately from microeconomic ones.

6. The rich and poor differ greatly in their direct dependence on biodiversity and the services of ecosystems.

7. There are no purely *economic* guidelines for choosing a discount rate. Responsibility to future generations is a matter of ethics, best guesses about the well-being of those in future and preserving life opportunities.

8. In general, a higher discount rate applied to specific cases will lead to the long-term degradation of biodiversity and ecosystems. A four per cent discount rate implies that biodiversity loss 50 years from now will be valued at only one-seventh of the same amount of biodiversity loss today.

9. A critical factor in discounting is the importance of environmental draw-down (destruction of natural capital) to estimates of g (as GDP growth). Is the current generation living on savings that should be passed to their descendents?

6 Identification of knowledge gaps and limitations and mapping into national policies: Challenges and options

The assessment of evidence on the relationship between ecosystem services and biodiversity helps not only in identifying the policy-relevant insights but also in filling the gaps in understanding the science which is critical for economic analysis. This can further be carried forward for research and must be remembered while designing policies for the national and global decision makers. Some of the emerging questions that are very relevant for economic analysis of change in ecosystems and biodiversity include:

6.1 Links between biodiversity, ecosystems and resilience

[nl]i) What are the roles of species interactions and functional diversity for ecosystem resilience?

ii) What are the drivers behind loss of resilience and how do they interact across scales?

iii) What are the impacts of climate and related environmental changes on ecosystem functioning through effects on species (re)distribution, numbers and process rates?

6.2 The dynamics of ecosystem services

i) How can we better quantify effects on regulating ecosystem services of an increase in non-sustainable use of provisioning services?

ii) What tools can contribute to accurate mapping of land and seascape units in terms of functioning and service provision?

iii) What specific tools could contribute to better assessment of spatial and temporal dynamics of service provision, especially in relation to defining who benefits, where and to what extent?

6.3 Understanding the dynamics of governance and management of ecosystems and ecosystem services

[nl]i) If all ecosystem services are taken into account, what is the appropriate balance between 'more diverse landscapes generating bundles of ecosystems services' and more intensively managed ecosystems like monocultures for food production?

ii) What are the trade-offs and complementarities involved in the provision of bundles of ecosystem services, and how do changes in the configuration of ecosystems affect their value?

iii) What are the most effective mechanisms for the governance of non-marketed ecosystem services, and how can these be designed so as to exploit future improvements in our understanding of the relationships between biodiversity, ecosystem functioning and ecosystem services?

6.4 Valuation and benefit transfer method

i) Since marginal values are likely to vary with ecosystem characteristics, socio-economic characteristics of beneficiaries, and ecological context, care needs to be taken to adjust transferred values when there are important differences between study and policy sites.

ii) It should be noted that the market size and rate of distance decay is likely to vary across different ecosystem services from the same ecosystem. It is also important to account for differences in site context in terms of the availability of substitute and complementary ecosystems and services.

iii) In cases where a high quality primary valuation study is available for a study site with very similar characteristics to the policy site, the unit transfer method may produce the most precise value estimate. In cases where no value information for a closely similar study site is available, value function or meta-analytic function transfer provide a sound approach for controlling for site specific characteristics.

iv) Aggregation of transferred unit values across the relevant population or ecosystem area needs to be undertaken carefully to avoid double counting values or misspecifying the market size for an ecosystem service.

v) Future valuation practitioners of biodiversity and ecosystem services should make explicit the procedures and methods used in their studies as well as openly acknowledge any obstacles that they may have encountered.

6.5 Valuations and its context

Social and institutional analyses suggest that valuation is essentially a matter of choosing how to perceive the human being itself, how to perceive human's place in Nature and how to perceive Nature itself. This is because the way we perceive our natural environment determines the way we value and change it. One way of incorporating a multilayered understanding of human–environment relations, and understanding the value and motivational linkages between the two, is to address the large gap that exists between the language in which the preference of the people for ecosystem services is elicited and the language in which people feel more at home. The languages of research and policy show similar dissonance. The more the discourse moves away from the common lives and real-life concerns to abstruse quantification and reductionism, the more people are likely to give preferences that are fudged and confused as much as these are confusing, merely because the choices we offer are far from adequate (Kumar and Kumar, 2008, p814). Valuation approaches aiming at addressing complex socio-ecological systems require attention to the challenge of understanding problems of credibility, saliency and legitimacy at the intersection of different knowledge systems and access to information at different levels and by different groups (Cash et al, 2006). In this sense, valuation mechanisms should be seen as part of a broader range of diagnostic and assessment tools and political–institutional mechanisms that facilitate the understanding of complex socio-ecological systems (Ostrom, 2009), as well as coproduction, mediation, translation and negotiation of information and knowledge within and across levels (Cash et al, 2006; Brondizio et al, 2009). The main lesson that comes across when one reviews valuation literature is to avoid a 'one size fits all' approach, or as Ostrom (2007) puts it when proposing a framework for the analysis of complex social-ecological systems, we need to move beyond panaceas.

Regulating services provide value through their role in assuring the reliability of service supply over space or time; sometimes expressed in terms of the resilience of the system to environmental shocks. That is, they moderate the variability or uncertainty of the supply of provisioning and cultural services. While an increase in biodiversity may increase production, experimental data indicate that for given environmental conditions this effect is small. The productivity of some biologically diverse communities, for example, has been found to be about ten per cent higher than the productivity of monocultures, but the effect often saturates at fewer than ten species. If environmental conditions are not constant, however, the effect may increase with the number of species providing that they have different niches and hence different responses to disturbances or changes in environmental conditions. For instance, the regulation of pest and disease outbreaks is affected by food web complexity.

i) Since people care about the reliability (or variability) in the supply of these services (people are generally risk averse), anything that increases reliability (reduces variability) will be valued. The value of regulating services accordingly lies in their impact on the variability in the supply of the provisioning and cultural services. An important factor in this is the diversity of the functional groups responsible for the services involved. The greater the specialization or niche differentiation of the species within a functional group, the wider the range of environmental conditions that group is able to tolerate. In some cases greater diversity with a functional group both increases the mean level and reduces the variability of the services that group supports. Indeed, this portfolio effect turns out to be one of the strongest reasons for maintaining the diversity of functional groups.

ii) The general point here is that the value of ecosystem components, including the diversity of the biota, derives from the value of the goods and services they produce. For each of the ecosystem services described in this chapter we have identified its sensitivity to changes in biodiversity. If greater diversity enhances mean yields of valued services it is transparent that diversity will have value. However, it is also true that if greater diversity reduces the variance in the yield of valued services that will also be a source of value. Since people prefer reliability over unreliability, certainty over uncertainty, and stability over variability, they typically choose wider rather than narrower portfolios of assets. Biodiversity can be thought of as a portfolio of biotic resources, the value of which depends on its impact on both mean yields and the variance in yields.

iii) It follows that there is a close connection between the value of biodiversity in securing the regulating services, and its value in securing the resilience of ecosystems. Since resilience is a measure of the capacity of ecosystems to function over a range of environmental conditions, a system that is more resilient is also likely to deliver more effective regulating services.

6.6 From micro foundation to macro policy

One of the major criticisms of economic valuation of biodiversity and ecosystem services is that most valuation exercises do not allow for ecosystems and economies to impact each other simultaneously. Also, although not necessarily a criticism, the frame of reference for most economic analysis of ecosystems is at the firm, household or individual industry level. Setting the analysis at the firm, household or industry level masks potential spillovers associated with actions taken in one sector of an economy on other sectors, and the corresponding impacts on macroeconomic variables like gross domestic product (GDP), aggregate savings rates or trade patterns. Similarly, the structural changes accompanying economic growth (sectoral composition), trade and consumption patterns can have far reaching impacts on the health and condition of ecosystems. Two examples of such impacts are: (i) the intensification of agricultural production accompanying economic growth, and its impact on soil salinity and water logging, and on genetic diversity; and (ii) regional and national subsidies to fish harvesting and the corresponding impacts on fish stocks and marine biodiversity.

There are other relevant examples where a macroeconomic framework could provide useful insights into how to better manage an ecosystem. One current issue in the economics of coastal ecosystems is the impact of economic growth and the increased conversion of mangrove forests into agricultural or aquacultural uses, and the corresponding impact on the ability of coastal ecosystems to support fish populations. Another issue is the concern with delta ecosystem destruction due to decreased – sometimes non-existent – river flow. With terrestrial ecosystems, an issue that has received much attention, and will continue to do so in the near future, is carbon sequestration. A simplistic summary of the current discussion is Southern countries have potential to serve as carbon sinks, but deforestation in these countries provides them with a significant source of GDP. A natural question to ask is, if Northern countries want a Southern country to serve as a carbon sink how much income would that country forgo if it decreased its desired rate of deforestation, or stopped it altogether? The answer to this question likely sets the lower limit on the North’s offer price to the South country to steward its forest assets differently.

Thus far, the thrust of this TEEB book has been to summarize the dominant conceptual frameworks and related methodologies used to measure the economic costs and benefits of ecosystem management policies. Such policies range from taxes to eliminate externalities in an existing prevailing market (market failures) to policy-induced distortions implemented at regional, national and global levels (e.g. tariffs to protect import-competing sectors or taxes on factors used by sectors that damage ecosystem health). The maintained hypothesis, here, is policy design can be better informed by properly using economic valuation and accounting exercises (with the understanding that policy rankings can be influenced by the discount rate used in the analysis).

The methodologies discussed thus far, however, do not adequately accommodate the interdependencies among different economic subsectors and ecosystem services. The methodologies also are not designed to measure the potential impact of a policy on the entire economy and its underlying ecosystem. The dependence of conventional economic sectors on ecosystems arises not only through the use of tangible factors like timber, water and fish, but also the use of intangible services like waste minimization, climate regulations or the control of vector-borne diseases (MA, 2003, 2005). Each of these services adds value to human well-being, but the ability of ecosystems to provide such services can be influenced by human behaviour.

The economics of tropical mangrove forests illustrate this point. Marine scientists have established a link between mangrove area and the carrying capacity of fish stocks – loosely speaking, the larger the area planted to mangrove forests, the larger the stock of fish the coastal ecosystem can support (see Barbier, 2003, 2009; Hoanh et al, 2006). The simplest economic story embedded in the relationship between mangrove forest area and fish stock is: the smaller the mangrove forest, the more costly it should be to harvest fish. Ignoring the land-conversion process, a natural question to ask is how will net returns to the fishing industry change as mangrove area falls? In answering this question, one might rightfully ignore the rest of the economy – especially if the fishing area contributes little to the overall economy. Alternatively, consider an economy wherein harvested fish are sold to firms who employ capital and labour to process fresh fish, who in turn compete with manufacturing and service sector firms for capital and labour. In this case, a decrease in mangrove area leads to an increase in fish harvesting costs, which in turn decreases the supply of fresh fish to processors. This can lead to higher fresh fish prices, which can place downward pressure on the demand for capital and labour by fish processors. These interactions can have implications for regional wages and rates of return to capital and investment patterns. Empirical examples (non-fishery) in Roe et al (2009would suggest the economic impact of a declining mangrove forest on both the fishing fleet and processing sector will depend on how important capital is relative to labour in producing each sector's output.

The two mangrove examples illustrate that some problems in ecosystem economics are well suited to microeconomic analysis, while others are not. Other examples will reveal that some issues can be examined without explicitly accounting for biophysical/ecosystem dynamics, while others should not. Insights into the economics of some issues can be garnered using a simple, single-sector growth model integrated with a biophysical model (e.g. the Dynamic Integrated Climate Economy (DICE) model developed by Nordhaus and Yang, 1996). On the other hand, a careful investigation of the economics of the mangrove–fishery problem in the previous paragraph likely requires integrating fishery dynamics with a dynamic economic model having multiple sectors.

Another issue which must be addressed is the relationship between the relative scale of economic activities and the natural ecosystem to which the economic system is a subset. Accounting for scale effects is critical in understanding how to manage an ecosystem and economy in a sustainable fashion. This management challenge is directly related to the concept of *the carrying capacity of Nature* (Arrow et al,, 1995). Just as the micro units of the economy – e.g. individual households and firms – function as part of a macroeconomic system, the aggregate economy functions as part of a larger system which Daily (1991) refers to as the natural ecosystem.

In response to these omissions, developing a conceptual framework that captures the economics of interactions between large-scale ecosystems and regional or national economies would facilitate the design of regional and national ecosystem and biodiversity policy. Take for example, policy suggested in TEEB in national policy, chs 2, 4 and 6). The economic theory underlying such policy could benefit by having its roots in dynamic, general equilibrium models. Even more importantly, the model should be empirically implementable. A good starting point for such a model might be found by linking the multi-species, dynamic general equilibrium ecosystem model (GEEM) discussed in Tschirhart (2009), Eichner and Tschirhart (2007) and Finnoff and Tschirhart (2008) with the dynamic, multisector macroeconomic models presented in Roe et al (2009). Such a framework could serve as a nice point of departure for examining the relationship between ecosystem management and (i) economic growth, (ii) structural transformation and (iii) simple trade and balance of payment issues. The Roe et al approach, however, may not be of much value in understanding the relationship between monetary policy and ecosystems.

A multisector, general equilibrium is desirable for two reasons. First, a general equilibrium model provides us with a direct means of understanding the interdependencies among economic subsectors and between these sectors and an ecosystem. Second, we want two or more sectors because policy makers are typically interested in knowing (or having an idea of) the likely impact of a policy on different stakeholder groups, for example manufacturing and agricultural lobbies. The desired framework is dynamic because both ecosystems and economies are dynamic entities, and it is important to understand the short- and long-run impact of a policy on an economy and the ecosystem with which the economy is linked. Furthermore, a long-run view is essential when delving into questions of sustainability. Finally, another desired feature of a model linking economics and the ecosystem would be for the model to accommodate ecosystems defined by two or more natural assets (e.g. the stock of coral reef serving as habitat for a stock of fish). We feel this is important because ecosystems are typically viewed as the result of interplay between two or more species or, more generally, between abiotic factors and biotic organisms.

Several challenges emerge when introducing dynamics into an economics–ecosystem model. For instance, standard approaches to empirical economic dynamic modelling does not typically accommodate hyperbolic or other potentially more complex discounting concepts. In addition, one might need to entertain the possibility of ‘shallow-lake’ dynamics, which can lead to multiple steady state equilibria (see Mäler et al, 2003; Heijdra and Heijnen, 2009). The model might also need to accommodate transboundary problems, which to address properly in an empirical fashion is quite challenging.

6.6.1 *Ecosystem Accounting*

Two issues linked with natural assets are: (i) the role of natural assets in measuring social welfare and sustainability, and (ii) the role of natural assets in social accounting and input–output tables – often referred to as 'green accounting'. Welfare, its definitions and measurement is the topic of interest to a wide spectrum of disciplines (see Stiglitz et al, 2009). For the past few decades, per capita gross domestic product (GDP) and per capita gross national product (GNP) have been popular indices of welfare. GDP is the total value of all final goods and services produced in a region, and GNP is GDP plus income earned by domestic citizens abroad less income earned by foreign citizens in the region. While GNP is a reasonable measure of economic activity, when measuring welfare in a dynamic setting (e.g. an evolving economy) some economists prefer using wealth or net national product (NNP) as a measure of economic well-being or social welfare. Here, wealth is typically viewed as the real value of stocks, and NNP is equal to GNP less depreciation [see Weitzman (1976); Dasgupta and Mäler (2000); Heal and Kriström (2005); Dasgupta (2009)].

Note, however, that these income and wealth based measures of welfare have well known problems. One problem with per capita GDP is it glosses over issues of distribution, and implicitly assumes the marginal valuation of an additional unit of income is the same for an impoverished person as it is for a rich person. Another problem is that traditional measures of GDP (and NNP) do not account for the depreciation (i.e. degradation) of environmental assets, biodiversity or ecosystems. Creation of the Human Development Index – where GDP is combined with health status and education levels – is an improvement in welfare measurement, but is however silent on natural capital's and ecosystems' contribution to human welfare (Dasgupta, 2009). See Stiglitz et al (2009) for a discussion of economic welfare: definitions, measurement and statistical issues, and data needs. The importance to traditional agriculturalists and subsistence farmers of ecosystem services from forest biomes (eg : freshwater and soil nutrient cycling, provision of fuelwood and fodder, mitigation of flood and drought damage to crops and property, etc) has been evaluated by TEEB and found to be a very material component of their household incomes (see “TEEB and Policy Making”). Traditional measures of economic performance are also insensitive to such equity and poverty implications of ecosystem services and of their ongoing losses.

The welfare discussion alluded to above is intimately related to the notion of sustainability. Although the issue of sustainability was taken up by Hicks and Lindahl beginning in the 1930s (Heal and Kriström, 2005), and discussed by Solow (1992 the potential long-run impact of environmental damages like global warming and concerns with the future availability of natural resources has more recently resurrected the sustainability discussion. Roughly speaking, an economy is on a sustainable trajectory if real wealth [Dasgupta and Mäler (2000)] or real income [Hicks (1939)], does not fall over time.

The national income account (NIA) is a fundamental macroeconomic indicator, which shows the level and performance of economic activities in the economy. The System of National Accounts (SNA) of the United Nations attempts to provide a benchmarked framework for measuring and summarizing national income data across all activities within an economy and to facilitate comparing such data across countries (SEEA 2003. A crucial component of the SNA is the estimation of GDP, where, at given market prices, the gross value of all the goods and services produced within an economy is estimated.

In measuring GDP, the contributions of ecosystem services such as bioremediation by wetlands, storm and flood protection by mangroves, and the prevention of soil erosion by forests are ignored. Hence, many ecosystem services which have a welfare-enhancing roles do not enter into macroeconomic value calculations. This is due to their relatively low perceived value as these resources and their contributions often fall outside the domain of the market, and hence remain un-priced and under-valued. Ultimately, this means that GDP underestimates the actual level of social welfare because it does not account for contributions from natural resources.

Economists are now in agreement that instead of measuring GDP or income – a flow concept – the more comprehensive measurement of the stock of wealth that includes the value of natural capital is a more meaningful and correct approach (Dasgupta and Mäler, 2000; Arrow et al, 2004; Dasgupta, 2009). The World Bank has attempted to implement the concept of ecosystem accounting, and created two reports in late 1990s – namely 'Monitoring Environmental Progress' (World Bank, 1995) and 'Expanding the Measure of Wealth' (World Bank, 1997). These reports are fundamentally based on Pearce and Atkinson's (1993) notion of wealth measurement and savings. Another important milestone in estimating natural wealth occurred in 2006, when the World Bank published 'Where Is the Wealth of Nations' a report that presents 'wealth accounts' of more than 100 countries (World Bank, 2006).

To date, there is a general consensus that natural assets are stocks that should enter a social accounting matrix (SAM) or input–output (IO) table as a factor account, and the provisioning and regulating services, if measurable, should enter as an intermediate input. Green accounts are a crucial ingredient in empirical general equilibrium modeling exercises because the SAM is the link between the macroeconomic theory and its empirical analogue. Although most countries have SAMs for one or more years, very few countries have SAM data that includes natural asset entries (Heal and Kriström, 2003).

Given the myriad of policy issues, it is useful to consider two broad categories of policy. One is related to the economic forces associated with the natural evolution of an economy (e.g. capital deepening and increased labour productivity), and how these forces impact (spill over onto) the exploitation of natural assets and ecosystems. In this category, policy instruments include taxes, subsidies, quotas, licences and property rights that are linked directly to natural assets and ecosystems. Similar instruments linked to other sectors can have indirect impacts on natural assets and ecosystems. The other set of policies are domestic macroeconomic policies (e.g. monetary policy) that cause major fiscal and trade imbalances, which in turn distort domestic product and factor markets. Similarly, foreign policies can have spillover effects on the home country.

Characteristics of a growing economy include capital deepening – here, broadly defined to include human capital – and typically an increase in the service sector's (housing, utilities, transportation, professional, banking and retail services) share of GDP, with most of the service goods not traded in international markets. Also, as economies grow (i) the share of the work force in agriculture falls as wages increase, which induces a substitution of capital for labour in production, and (ii) household expenditures on services and entertainment goods increase.

The possible impacts a successful economy can have on ecosystems and biodiversity 'works in reverse' for an unsuccessful, slow-growing economy. Often, poor economic performance is also associated with economic crises and volatility in factor incomes. Invariably, these conditions tend to slow the exodus of labour out of agriculture, slow the rate of capital deepening and dampen a country's incentives and political will to prevent the degradation of natural and environmental resources. Economic crises often result when countries pursue policies that lead to fiscal deficits or external debt that make their economies vulnerable to economic shocks. The needed within-country-adjustments – even with the assistance of international agencies and friendly governments – is invariably a decrease in government spending, an increase in transfers from households either directly through taxes and fees, and costs of adjustment as resources are reallocated from the production of home goods to the production of internationally traded goods. Until the imbalances are corrected and debt obligations met, household real disposable income is typically much less than before the shock, with wage income suffering the greatest decline. Ecosystem exploitation (e.g. harvesting timber for cooking and fuel) is exacerbated by an incentive to increase exploitation by those whose wages and employment have fallen, while at the same time, government budgets for the management of these resources is even more constrained.

Finally,

Through the TEEB series including in this book, in 'TEEB for Policy Makers' (TEEB in National Policy, 2011) and 'TEEB for Local Governments' (TEEB in Local Policy, 2011), we follow a tiered approach in analysing problems and suitable policy responses. We find that at times, it suffices just to *recognize* value - be it intrinsic, spiritual or social - to create a cost effective policy response for conservation or economically sustainable use of biodiversity and ecosystem services. At other times we may need to *demonstrate* economic value in order for policy makers to respond - such as a wetland conserved near Kampala (see TEEB in Local

Policy, 2011, Chapter 8) for its waste treatment function instead of reclaiming it for agriculture, or river flats maintained for their many ecosystem services instead of converting them for residential use near New Delhi (see TEEB in National Policy, 2011, Chapter 10, Box 10.6).

It is not a "risk-free" exercise to demonstrate value by deriving and propagating shadow prices. There is always the risk that misguided policy-makers or exploitative interests may want to use these prices for the wrong ends. Therefore, our proposition is that the act of valuing the flows and stocks of Nature is ethically valid so long as the purpose of that exercise is, first and foremost, to demonstrate value in order to instigate change of behaviour, and to pre-empt damaging trade-offs based on the implicit valuations that are involved in causing the loss of biodiversity and degradation of ecosystems.

TEEB has also focused considerably on changes which *capture* value by rewarding and supporting good conservation through a variety of means – Payments for Environmental Services (PES) , IPES, establishing new markets, EHS reforms, etc. (see TEEB in National Policy, 2011, Chapters 5 and 9, and TEEB in Local Policy, 2011, Chapter 8). These instruments offer a mechanism to translate external, non-market values of ecosystem services into real financial incentives for local actors to provide such services (Ferraro and Kiss, 2002). There is also a need to use existing knowledge and examples of success to develop more effective governance institutions, including property rights regimes and regulatory structures.

Recognizing that biodiversity underpins the foundations of economies, and of human well-being, is one thing. Translating that knowledge into concrete changes that will influence behaviour for the better is another, formidable challenge. It is one that must be met if the failures of the recent past are not to be repeated and compounded, with the potential for ever-increasing human and financial costs. This TEEB book and other forthcoming volumes (especially TEEB in National Policy, 2011, and TEEB in Local Policy, 2011) make the case that greater economic and ecological rationality in addressing natural capital is not only necessary but possible, and indeed, that it is amply evidenced in many instances which deserve more attention, investment, and opportunity to replicate and to scale into wider use around the world.

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