Chapter 5

The economics of valuing ecosystem services and biodiversity

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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.

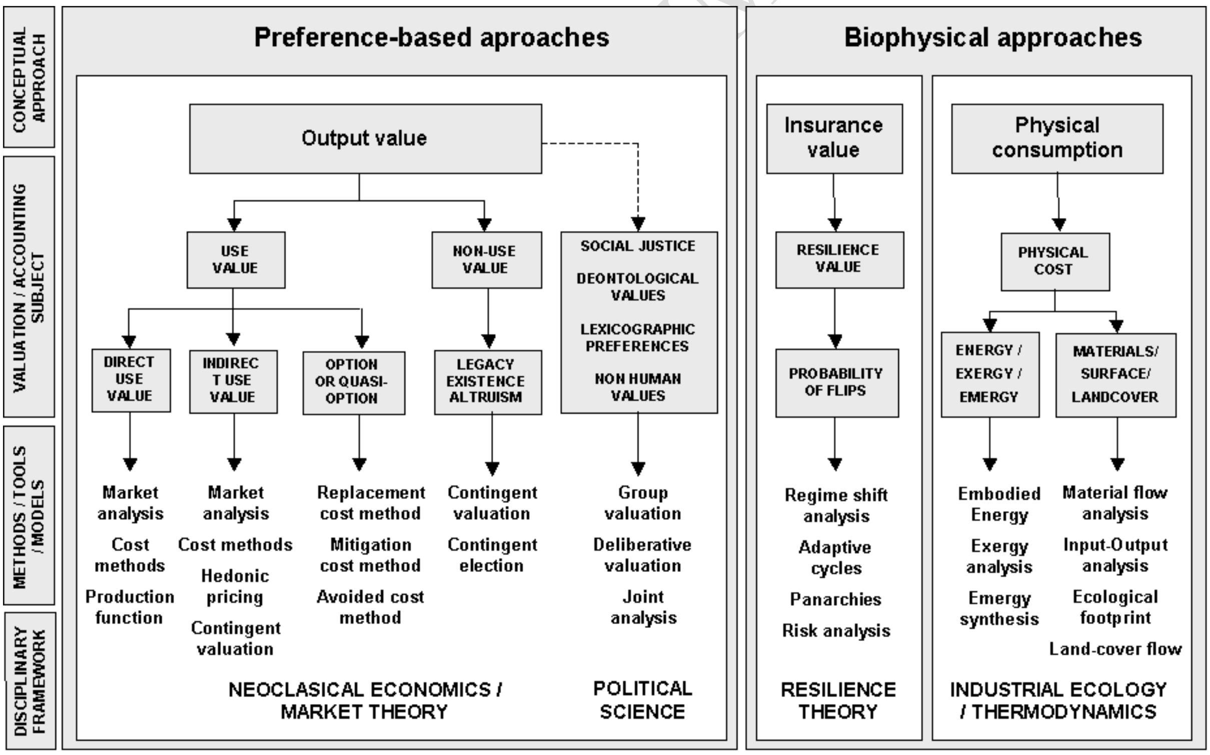
2.2 Valuation paradigms

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).

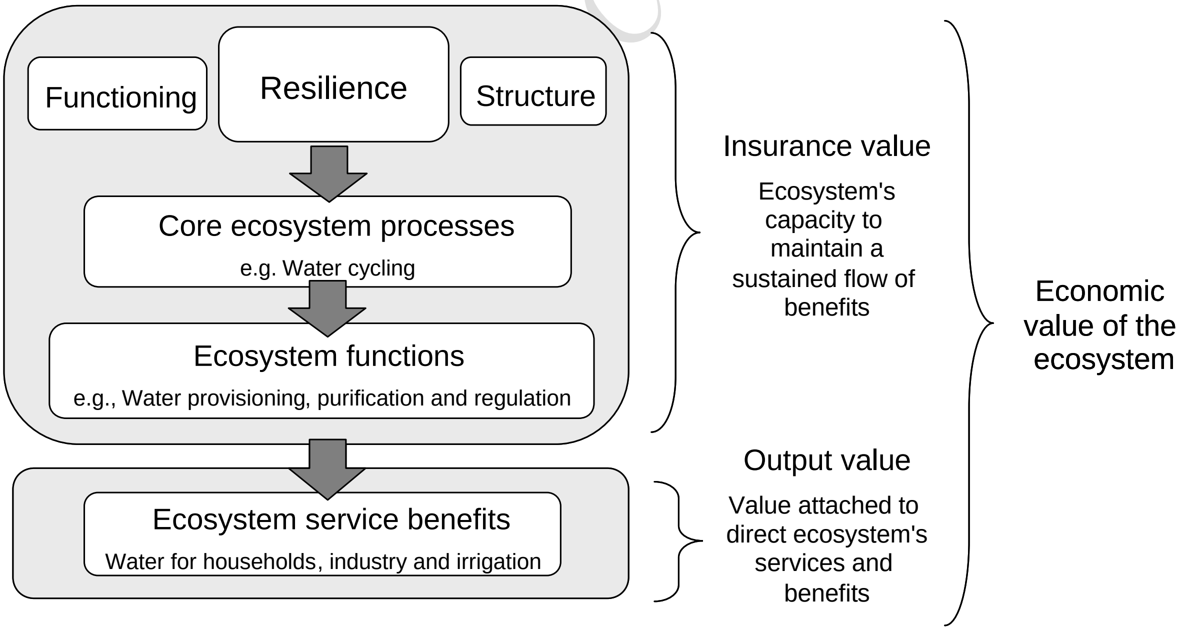


Figure 2: Insurance and output value as part of the economic value of the ecosystem
The figure poses *insurance value* (related to the ecosystem's resilience and *output value* (related to ecosystem service benefits) as the two main components of the economic value of the ecosystem.

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).

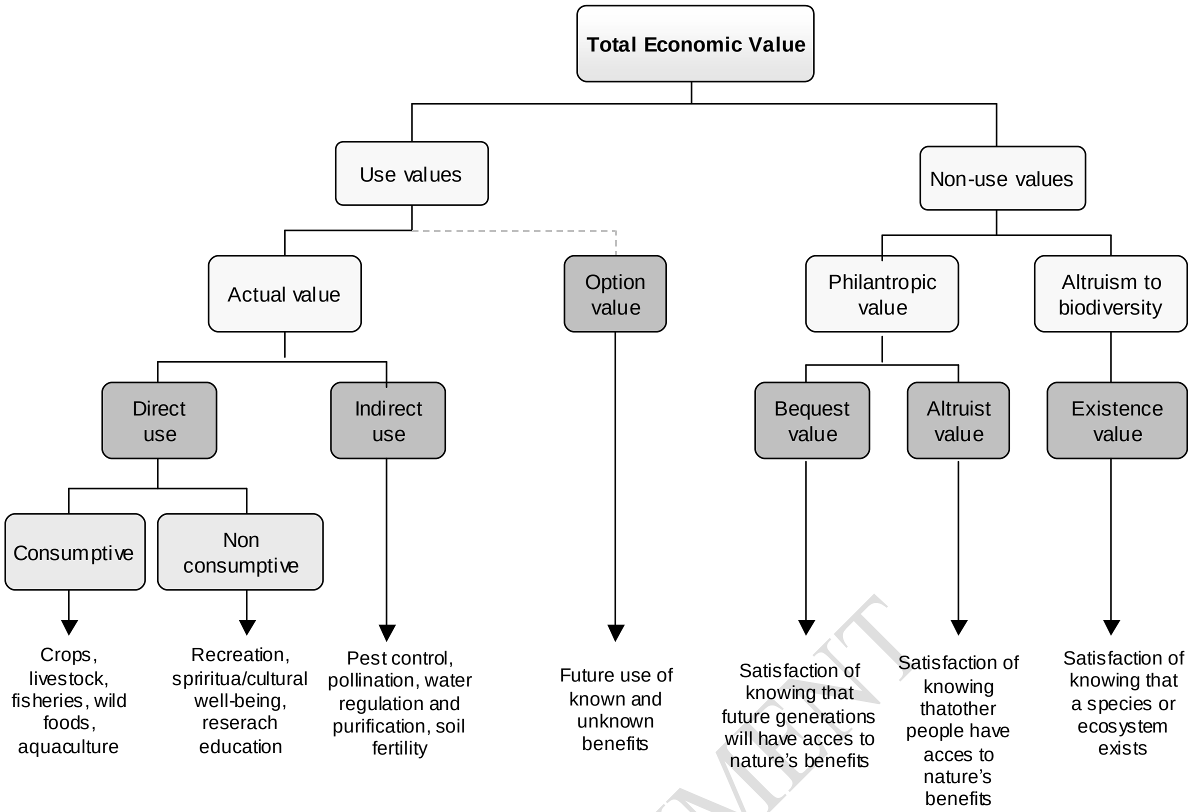


Figure 3: Value types within the TEV approach

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

Approach		**Method**	**Value**
Market valuation	Price-based	Market prices	Direct and indirect use
	Cost-based	Avoided cost	Direct and indirect use
		Replacement cost	Direct and indirect use
		Mitigation / Restoration cost	Direct and indirect use
	Production -based	Production function approach	Indirect use
		Factor Income	Indirect use
Revealed preference		Travel cost method	Direct (indirect) use
		Hedonic pricing	Direct and indirect use
Stated preference		Contingent Valuation	Use and non-use
		Choice modelling/ Conjoint Analysis	Use and non-use
		Contingent ranking	Use and non-use
		Deliberative group valuation	Use and non-use

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.

<table>
<tr><th colspan="3">Method</th><th>Comment /example</th><th>References</th></tr>
<tr><td rowspan="5">Market valuation</td><td colspan="2">Market Price</td><td>Mainly applicable to the “goods” (e.g. fish) but also some cultural (e.g. recreation) and regulating services (e.g. pollination).</td><td>Brown et al. 1990; Kanazawa 1993</td></tr>
<tr><td rowspan="3">Cost based</td><td>Avoided cost</td><td>The value of the flood control service can be derived from the estimated damage if flooding would occur.</td><td rowspan="3">Gunawardena & Rowan 2005; Ammour et al. 2000; Breaux et al. 1995; Gren 1993</td></tr>
<tr><td>Replace-ment cost</td><td>The value of groundwater recharge can be estimated from the costs of obtaining water from another source (substitute costs).</td></tr>
<tr><td>Mitigation/ restoration costs</td><td>E.g. cost of preventive expenditures in absence of wetland service (e.g. flood barriers) or relocation.</td></tr>
<tr><td colspan="2">Production function / factor income</td><td>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.</td><td>Pattanayak & Kramer 2001</td></tr>
<tr><td rowspan="2">Revealed preferences</td><td colspan="2">Travel Cost Method</td><td>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.</td><td>Whitten & Bennet 2002; Martín-López et al. 2009b</td></tr>
<tr><td colspan="2">Hedonic Pricing Method</td><td>For example: clean air, presence of water and aesthetic views will increase the price of surrounding real estate.</td><td>Bolitzer & Netusil 2000; Garrod & Willis 1991</td></tr>
<tr><td rowspan="3">Simulated valuation</td><td colspan="2">Contingent Valuation Method (CVM)</td><td>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.</td><td>Wilson & Carpenter 2000; Martín-López et al. 2007</td></tr>
<tr><td colspan="2">Choice modelling</td><td>It can be applied through different methods, which include choice experiments, contingent ranking, contingent rating and pair comparison.</td><td>Hanley & Wright 1998; Lii et al. 2004; Philip & MacMillan 2005</td></tr>
<tr><td colspan="2">Group valuation</td><td>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.</td><td>Wilson & Howarth 2002; Spash 2008</td></tr>
</table>

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.

Row Labels	Forests				Forests Total	Wetlands				Wetlands Total	Grand Total
	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).

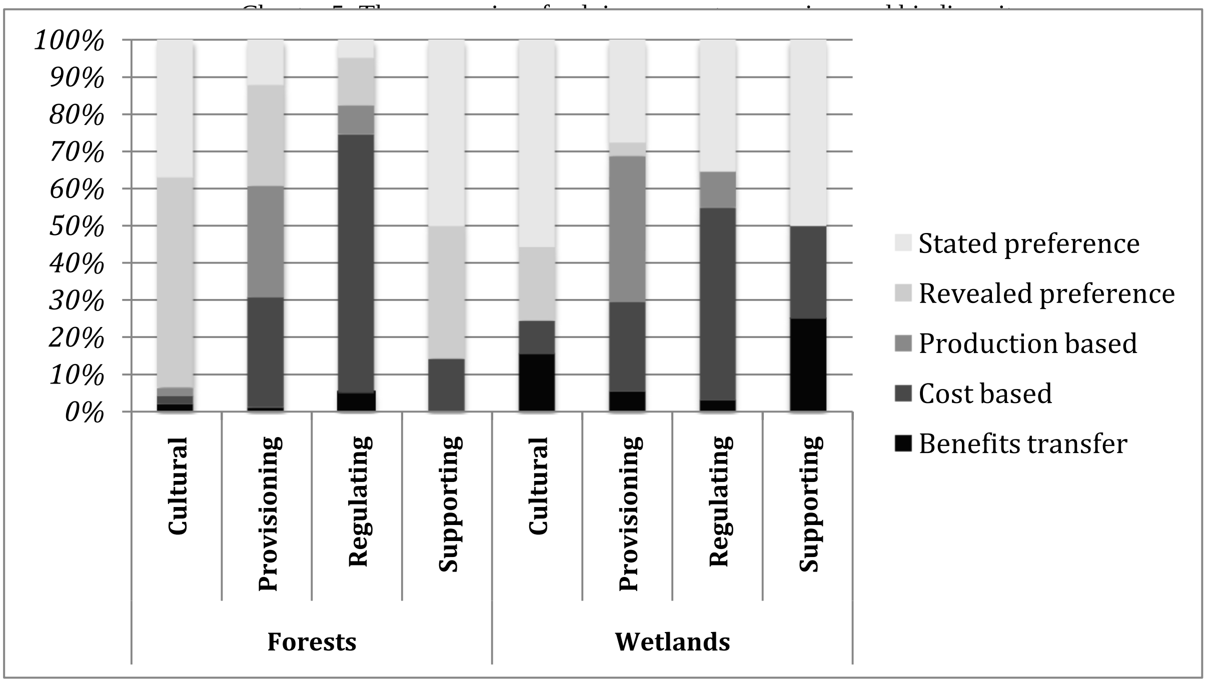


Figure 4: **Vauation approaches that have been used to value ecosystem services provided by forests and wetlands**

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., *2*004).

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.

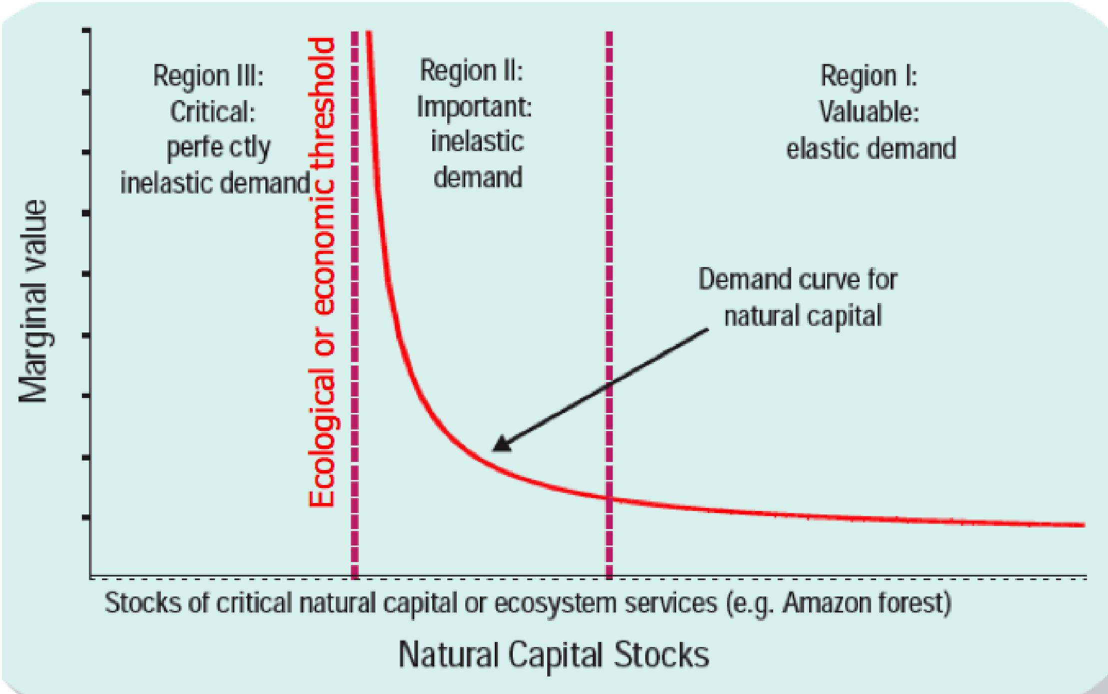


Figure 5. **The demand curve for natural capital** (Farley, 2008) Figure 5 depicts a stylized demand curve for critical natural capital with an economic or ecological threshold.

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. Eijja 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 *(na-tural 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 inspiration *(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)*	***Stated preference*** ***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;)* ***Production based*** ***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***	***Stated preference*** ***CVM*** *(e.g. Costanza et.al., 1997)* ***Stakeholder Analysis and CVM*** *(e.g. Bhatta, 2000)* ***Cost based*** ***Restoration cost*** *(e.g. Emerton, 2005)*	***NA***	***Stated preference*** ***Contingent Ranking*** *(e.g Lynam et al. 1994)* ***CVM*** *(e.g. Gunawar-dena et al. 1999; Shaikh et al. 2007; Loomis 1992)* ***Revealed preference*** ***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)* ***Production based*** ***Factor Income*** *(e.g. Peters et al. 1989; Hodgson & Dixon 1998; Carret & Loyer 2003; Anderson 1987; Mäler 1992; Moskowitz &Talberth 1998; Verma, 2008)* ***Cost based*** ***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) ***Cost based*** ***Replacement cost*** *(e.g. Gren, et al. 1994; Abila 1998)* ***Benefits transfer*** ***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)*	***Stated preference*** ***Choice modelling*** *(e.g. Gordon et al. 2001)* ***Participatory Valuation*** *(e.g. IUCN-WANI 2000)* ***Revealed preference*** ***Public Investments*** *(e.g Powicki 1998 ; Emerton, 2005)* ***Production based*** ***Factor Income*** *(e.g. Emerton, 2005)* ***Cost based*** ***Opportunity cost*** *(e.g. Emerton 2005)* ***Replacement cost*** *(e.g. Gren et al., 1994)* ***Restoration cost*** *(e.g. Verma, 2001)*	***NA***	***Cost based*** ***Restoration cost*** *(e.g. L.Emerton, 2005)*	***NA*** ***CVM for non-user benefits*** *(e.g. James and Murty, 1999;*	***Revealed preference*** ***TCM*** *(e.g. Wittington et al. 1990, 1991)* ***Production based*** ***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)* ***Cost based*** ***Avoided cost*** *(e.g. Chaturvedi, 1993)* ***Treatment/ Mitigation cost*** *(e.g Kumari 1996)*	***NA***	***Stated preference*** ***CVM*** *(e.g. Kadekodi, 2000;)*	***NA***
3	***Raw Materials****(e.g. fibres, timber, fuelwood, fodder, peat,fertilizer, construction material etc.)*	***Stated preference*** ***Contingent ranking*** *(e.g. Emerton ,1996;)* ***CVM*** *(e.g. Hanley & Craig 1991)* ***Participatory Valuation*** *(e.g. Eaton, 1997; Emerton, 2005; IUCN-WANI, 2005;)*	***NA***	***Stated preference*** ***Participatory valuation*** *(e.g. Eaton, 1997)* ***Cost based*** ***Restoration cost*** *(e.g. Emerton, 2005)*	***NA***	***Stated preference*** ***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***	***Stated preference*** ***CVM*** *(e.g. Ninan and Sathyapalan, 2005;)* ***Cost based*** ***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; Pattanayak & 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)*		***<u>Stated preference</u>*** ***Choice Modelling*** *(e.g. Carlsoon et al. 2003)* ***<u>Cost based</u>*** ***Replacement cost*** *(e.g. Gren et al. 1994)* ***<u>Benefits transfer</u>*** ***Benefits Transfer*** *(e.g. Andréassen Gren & Groth, 1995)*						
	CULTURAL								
18	***Aesthetic*** *(e.g. appreciation of natural scenery, other than through deliberate recreational activities)*	***<u>Stated preference</u>*** ***Choice modelling*** *(e.g. Bergland 1997)* ***CVM*** *(e.g. Mahan, 1997)* ***<u>Revealed preference</u>*** ***Hedonic pricing*** *(e.g. Verma 2000; Mahan, 1997)* ***<u>Cost based</u>*** ***Replacement Cost*** *(e.g. Gupta, 1975;)*	***NA***			***<u>Revealed preference</u>*** ***Hedonic pricing*** *(e.g. Garrod & Willis 1992; Tyrvaninen and Meittinen 2000; Kramer et al. 2003; Holmes 1997)* ***TCM*** *(e.g. Holmes 1997)* ***<u>Cost based</u>*** ***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)*	***<u>Stated preference</u>*** ***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***	***<u>Stated preference</u>*** ***CVM*** *(e.g. Desvousges et al. 1987)*		***<u>Stated preference</u>*** ***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***	***<u>Stated preference</u>*** ***Option value*** *(e.g. Walsh et al. 1984)* ***<u>Revealed preference</u>*** ***Expenditure on Wilderness***	***<u>Stated preference</u>*** ***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)* ***Revealed preference*** ***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)* ***Cost based*** ***Opportunity Cost*** *(e.g. Loomis et al. ,1989;)* ***Protection cost*** *(e.g. Pendleton 1995)* ***Replacement and Conversion Cost*** *(e.g. R. Abila,1998;)* ***Benefits transfer*** ***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)* ***Revealed preference*** ***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)* ***Production based***		*(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)* ***<u>Benefits transfer</u>*** ***Production*** ***Benefits transfer*** *(e.g. Walsh & Loomis 1989)*			
20	***Educational*** *(e.g. Opportunities for formal and informal education and training)*	***NA***				***<u>Revealed preference</u>*** ***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)*	***<u>Stated preference</u>*** ***CVM*** *(e.g. Maharana et al., 2000)*				***<u>Revealed preference</u>*** ***TCM & CVM*** *(e.g. Maharana et al., 2000)*			***<u>Stated preference</u>*** ***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*	***<u>Stated preference</u>*** ***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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