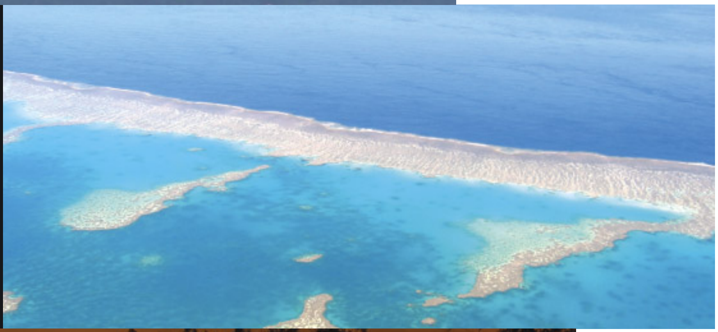
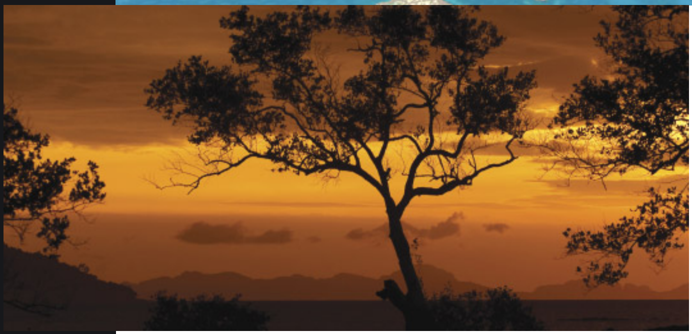
Secretariat of the Convention on Biological Diversity

CBD Technical Series No. 28

AN EXPLORATION OF TOOLS AND METHODOLOGIES FOR VALUATION OF BIODIVERSITY AND BIODIVERSITY RESOURCES AND FUNCTIONS





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An exploration of tools and methodologies for valuation of biodiversity and biodiversity resources and functions

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FOREWORD

Over the last decades, human beings have changed global ecosystems faster and more extensively than in any comparable period of time in human history, leading to an unprecedented and ongoing loss of biodiversity. The size of key natural ecosystems such as tropical forests or wetlands has already shrunk dramatically, or become increasingly fragmented, with disastrous results for biodiversity. Species are becoming extinct at 1,000 times the typical background rate—leading scientists speak about the sixth wave of extinction taking place in Earth's history. This is the main message of the Global Ecosystem Assessment, an assessment of the world's ecosystems which was prepared by 1395 experts from 95 countries.

According to the Assessment, the loss of biodiversity constitutes a concern for human well-being, especially for the well-being of the poorest. Biodiverse ecosystems provide essential resources and goods, such as food, fibre, and medicines. The ecosystem functions that support these provide other vital services, such as the regulation of water flows and levels, protection against extreme weather, the purification of air and water, the prevention of soil erosion, and opportunities for recreation and spiritual reflection. In its evaluation of these ecosystem services, the Assessment found that 15 out of 24 examined are in decline.

Most of these ecosystem services are not traded on markets and thus do not bear a price tag, making it difficult to make informed choices about their conservation and sustainable use. Absence of a price does not mean the absence of economic value however. Revealing the hidden value of ecosystem services through valuation techniques, in particular on non-market valuation, is an important mechanism for integrating biodiversity considerations in economic decision-making. Applications of improved valuation techniques can lead to interesting observations. For example, valuation techniques tell us that although many individuals benefit from activities that lead to biodiversity decline and the associated loss in ecosystem services, the costs borne by society of these activities are often higher. The annex of the present publication provides a number of compelling examples.

Since its inception, Parties to the Convention on Biological Diversity have expressed considerable interest in work on valuation, including the review of valuation information and research into appropriate and cost-effective valuation methodologies. The present document responds to a specific request of the eighth meeting of the Conference of the Parties, which took place in Curitiba, Brazil, on 20–31 March 2006, to disseminate information on valuation methods including through the CBD Technical Series, in order to promote a common understanding of valuation techniques among governments and stakeholders.

The Conference of the Parties at its eighth meeting underlined that the application of practical valuation methods can contribute to meeting the 2010 biodiversity target. The target commits Parties to the Convention to achieve, by 2010, a significant reduction of the current rate of biodiversity loss at the global, regional and national level. Achieving this target is ambitious, but vital. It is my hope that the present publication will provide useful technical background information to Parties, other governments, and stakeholders on the voluntary application of such valuation methods, and will hence be instrumental in achieving the 2010 biodiversity target, as a contribution to poverty alleviation and for the greater benefit of all life on earth.

Ahmed Djoghlaf
Executive Secretary of the Convention on Biological Diversity

I. INTRODUCTION AND ACKNOWLEDGEMENTS

The present document was initially prepared as an information document for the eleventh meeting of the Convention's Subsidiary Body on Scientific, Technical and Technological Advice (SBSTTA), which took place in November 2005, in Montreal, Canada. It responded to a request to the Executive Secretary expressed by the seventh meeting of the Conference of the Parties to the Convention, to explore, in cooperation with the Millennium Ecosystem Assessment, the Organisation for Economic Co-operation and Development and relevant international organizations, existing methodologies for valuation of biodiversity and biodiversity resources and functions, as well as other tools for prioritization in decision-making, by preparing a compilation of existing valuation tools that provides an overview of the discussion on their methodological status, if appropriate, as well as an assessment of their applicability in terms of effectiveness and capacity preconditions. The document subsequently served SBSTTA as a basis for the development of concise "Options for the application of tools for valuation of biodiversity and biodiversity resources and functions", which are available as a separate publication.

The first drafts of the body of the document were prepared by Dr. Dominic Moran, and the Secretariat gratefully acknowledges this contribution as well as his valuable support in the further development of the document. Thanks go also to Mr. Hans Haake, University of Oldenbourg, Germany, for his support in preparing the final draft as well as the compilation and synthesis of the valuation studies provided in the annex to the document. On the Secretariat level, the document was drafted and finalized by Dr. Markus Lehmann, economist.

The analysis and discussion of the valuation tools provided in section three of the document is to a considerable extent based on the review and assessment of valuation tools provided in chapter 2.3.3.1 of volume 1 of the Millennium Ecosystem Assessment, and the Secretariat wishes to extend its thanks to the Coordinating Lead Authors of this chapter, Dr. Ruth de Fries and Dr. Stefano Pagiola, as well as its Lead and Contributing Authors and reviewers.

Parties to the Convention on Biological Diversity, other Governments, and relevant international organizations and experts were invited to participate in a peer review of the document, and the Secretariat wishes to gratefully acknowledge the reviews that were provided by the following Parties: Argentina, Canada (three reviews), Egypt, European Community and its Member States, India, Kenya, Netherlands, Sri Lanka and Ukraine.

Comments were also provided by the United States of America, as well as by the Food and Agriculture Organization of the United Nations (FAO), the Organisation for Economic Co-operation and Development (OECD), and the United Nations Environment Programme (UNEP). Personal reviews were provided by Prof. Ronaldo Seroa da Motta, Research Institute for Applied Economics, Rio de Janeiro, Brazil; and by Dr. Renat Perelet, Institute for Systems Analysis, Russian Academy of Sciences, Moscow, Russian Federation.

The Secretariat wishes to gratefully acknowledge the financial support of the Government of the Netherlands for the publication of this document.

II. METHODOLOGICAL ISSUES IN VALUATION

A. DEFINING VALUE

Biodiversity as well as biodiversity resources and functions are intuitively valuable. Few would contest the fact that the decline of biodiversity would be costly to humankind, in particular with regard to those functions that cannot be replicated. But this general truth does not shed much light on how to identify, describe and measure the specific values that are held in respect of biodiversity and biological resources and functions.

The term value is used in different ways amongst a range of academic disciplines. According to the Oxford Dictionary, there are three main types of uses of the term "value": (i) *exchange value*, that is, the (relative) price of a good or service in the market; (ii) *utility*, that is, the use value of a good or service, which can be very different from the market price (e.g. the market price of water is very low, but its use value very high; the reverse is the case for, for example, diamonds or other luxury goods); and (iii) *importance*, that is, the appreciation or emotional value attached to a given good or service (e.g. the emotional or spiritual experience some people have when viewing wildlife or natural scenery, or our ethical considerations regarding the existence value of wildlife).

Different disciplines define and use these terms in different ways. In economics, value and utility are unambiguously anthropogenic.[1] For instance, in the case of marketed goods and services, it is humans who reveal value, in terms of their so-called willingness-to-pay, by the process of exchange. Similarly utility is derived by humans. Even the concept of importance is only meaningful if assigned by, and inferred from, human choices or decisions on behalf of other living organisms. But other disciplines may assign different interpretations to value or importance, which may or may not be linked to values ascribed by human beings. For example, anthropology may infer value from cultural norms and practices that are in some sense non-negotiable (e.g. sacred groves). Theologians and ethicists may base importance on moral or spiritual criteria that are neither observed nor measurable (but nevertheless strong motives), and may also point out that the predominant role of humans in utilitarian thinking displaces intrinsic value and the right of other species to exist. And last but not least, ecologists will be interested in the importance of attributes or functions of a system to maintain ecosystem resilience. This is an objective criterion, that is, irrespective of its relevance to humans.

In what follows it is important to bear in mind these disciplinary distinctions and the fact that different perspectives on value lead to differing views on the practicality of measurement and, by extension, use in policy making.

B. VALUATION

While there is growing awareness of the value and importance of diversity *per se*, there is a lack of consensus on how diversity can be defined and measured. For example, species richness is frequently the

1 A distinction is made in philosophy between anthropo*centric* and anthropo*genic* value. Something has anthropocentric value when it is good for a human subject. Foods and other goods used by humankind have anthropocentric (and instrumental) value. Anthropogenic value, on the other hand, is value that is attributed *by* a human subject, but not necessarily value *for* a human. So, for example, many people take an old growth forest to have value whether or not it is actually used or appreciated firsthand by a person. See Hiller 2005, Callicott and Baird 1999. Because of the concept of existence value, explained further below, the term anthropogenic is used here.

only accessible indicator of species diversity, although it is well known that a head count of the number of apparently different species in an area may not be a good proxy for the portfolio effect of genetic distance between them. Some context-sensitive index or set of indices of biodiversity change would be fundamental to any economic valuation of diversity. Indices could in theory be based on phylogenetic data. In practice this data is not readily available as a basis for prioritization. However, other prioritization devices, discussed below, employ non-monetary measures of value that may encompass genetic distance.

In consequence, valuation does normally not entail measuring the economic value of bio*diversity as such.*[2] Instead, valuation typically focuses on the economic values of the goods and services generated by biodiversity resources and/or functions—the so-called ecosystem services.[3] A comprehensive assessment of the values of ecosystem services[4] has recently been undertaken by the Millennium Ecosystem Assessment. The Millennium Ecosystem Assessment adopted a wide understanding of ecosystem services, which includes goods under the concept of "provisioning services". While this understanding departs from the usual economic distinction between "goods" and "services", it will be adopted in the remainder of the note for the sake of ensuring consistency with the terminology introduced by the Millennium Ecosystem Assessment.[5]

It is noteworthy that the term "economic" is to be understood in a broad sense. Based on welfare economics, economic valuation recognizes that individuals may assign value for different reasons or motives, and not only for the immediate benefits of commercial exploitations of resources (as a narrow interpretation of the term "economic" may suggest).

In a recent note, leading scientists under the Millennium Ecosystem Assessment clarified that, as used and defined in the Assessment and as already used in existing international documents, the term and concept of "ecosystem services" in no way implies an automatic requirement or obligation on the part of the consumer to pay directly for the supply of the service. The term does, however, imply that the service is of value to people (in terms of economic, health, cultural or other benefits) and that the degradation or loss of the service represents a harmful impact on human well-being. The note also highlighted the different policy options at hand for reducing the degradation of ecosystem services, and underlined that it is a matter of societal choice which option or options to use.[6]

2 Pearce and Moran, 1994; Pearce 2001. There is however some literature that seeks to determine the value of biodiversity components by the genetic composition of species rather than by the species themselves (see, for example, Polasky et al., 1993; Metrick and Weitzman, 1996; 1998; Weitzman, 1998). Empirical applications of these methods have been presented by Weitzman 1993 and Solow et al. 1993.

3 See for instance Daily and Dasgupta for an explanation of this concept. As the same functions often contribute to the production of different services, simply adding the values of different services would likely not produce an accurate estimate of the value of the underlying functions. See Pearce and Moran 1994.

4 See Millennium Ecosystem Assessment 2003. See also Christie 2004 as well as Daily and Dasgupta 2001 for further discussion, and Eftec 2005 for a literature review of the economic, social and ecological value of ecosystem services.

5 See also Daily and Dasgupta 2001 for a similar conceptualisation. The Millennium Ecosystem Assessment was carried out between 2001 and 2005 to assess the consequences of ecosystem change for human well-being and to analyze options available to enhance the conservation and sustainable use of ecosystems and their contributions to human well-being. Responding to requests for information including through the Convention on Biological Diversity, it was carried out by 1,395 experts from 95 countries, and has been extensively peer reviewed by governments and experts.

6 See Reid, V. et al. 2005. The note reacted to concerns raised in the context of the discussion on water pricing and the privatisation of water resources.

Economics generally assigns value on the basis of direct or indirect tradeoffs, that is, actions that show people making sacrifices in favour of specific goods and services, thus revealing their willingness-to-pay for these goods and services by exchanging them on markets. These actions can be explained by a robust theory of demand that posits specific axioms or rules about the consistency in which these choices are made. It is the consistency of the predictions of this theory that enables economists to infer what people value based on what they actually do.

Environmental economics has extended demand theory to goods and services that are not traded on markets, including most ecosystem services (which include goods according to the understanding of the Millennium Ecosystem Assessment). As they are not traded on markets, their value is not captured in market prices. The reason is that many ecosystem services bear characteristics of what economists call "public goods". One important characteristic of public goods is that nobody can be excluded from their use. For this reason, markets cannot spontaneously develop for public goods, and the value of these public goods will therefore not be reflected in a market price. This has also the consequence that the prices of many marketed goods and services will not adequately reflect the essential role of these services in their production, which, in turn, will lead to distorted decisions by consumers and producers. Public decision-making and its allocation of public funds will also be distorted if the repercussions of governmental activities on these biodiversity resources and functions, and the associated ecosystem services, are not adequately factored in.[7]

In consequence, undertaking valuation does not only raise awareness of the hidden benefits of biodiversity conservation in terms of maintaining critical ecosystem services. It has also the potential of improving public decision-making as well as, under specific circumstances, of improving legal decision-making.[8]

The proposals on the design and implementation of incentive measures, endorsed by the sixth meeting of the Conference of the Parties to the CBD, as far as they are consistent with Parties' national polices as well as their international obligations, underlined that valuation can also support the design of other incentive measures for the conservation and sustainable use of biodiversity.[9] It was recognized by the Conference of the Parties that incentive measures should not negatively affect biodiversity and livelihoods of communities in other countries. In that regard, valuation could also contribute. For instance, the valuation of the ecosystem services that are relevant under a given decision-making problem, at all relevant scales (local, regional and/or global, on-site and/or off-site), could contribute to ensure that repercussions on biodiversity at all scales are taken into consideration in decision-making.

7 De Groot et al. (2006), at page 11, show a more detailed overview of how "normal markets" fail to specifically reflect the value of wetlands.

8 See the discussion under IV B 4 below.

9 Decision VI/15, annex I, paragraph 22. It has been noted by the OECD that such calibration is of particular importance for those instruments that seek to directly correct prices, such as fees or direct payments for environmental services. In cases where property rights could be established on the relevant biodiversity assets, a market price would emerge endogenously. Even in the latter case however, valuation would still be useful to determine the magnitude of the policy problem and which policy instruments to choose.

10 Depending on the question that is to be investigated, focus is sometimes given to the so-called "willingness-to accept." For instance, if an area is to be protected and people who have the legal title to use that area would be no longer allowed to do so, they might be asked for how much compensation they are willing to voluntarily give up their right to use the area. Willingness-to-accept generally raises important problems with biases, which is why the concept of willingness-to-pay is generally preferred. See Hanemann 1991.

Since the 1960s, considerable efforts have been made by economists to develop methods that can elicit the "hidden" value of non-marketed natural resources. These methods use the aforementioned sacrifice or "willingness to pay",[10] based on actual or hypothetical behaviour, to infer the value of the resource. There are many reasons why people are indirectly observed to, or directly state that they are willing to, make tradeoffs between their endowment (in terms of time, labour effort, monetary income or wealth) and safeguarding non-marketed natural resources, including safeguarding specific levels of ecosystem services. The framework commonly used for describing the different types of economic value ascribed to natural resources is known as the Total Economic Value (TEV) and will be presented below.

Valuation usually attempts to measure the value of ecosystem services in monetary terms, in order to provide a common metric in which to express the benefits of the variety of services provided by ecosystems. This explicitly does not mean that only monetary sacrifices, or only services that generate monetary benefits, are taken into consideration. What matters is that people are willing to make tradeoffs. If the relevant people were for instance subsistence farmers, these tradeoffs could be initially measured by the labour time they are willing to provide for achieving some environmentally-friendly outcome. In order to have a common metric, this effort could then be transformed into a monetary figure by applying for instance the local or domestic wage rate.

The economics profession is divided on whether valuation is adequate or sufficient to deal with the more fundamental issues that are also involved in biodiversity management. It is in particular suggested that some biodiversity functions are key to the survival of global ecosystems including humans (the so-called life support function) and should therefore be treated as a fundamental constraint and not as an element of the set of possible economic choices.[11] Put another way, all economic choices must be made within some ecological constraints otherwise the global system may collapse. The standard toolbox of economic valuation is said to be of limited if any use for the identification of these global constraints. Alternative approaches such as setting a safe minimum standard may be more suitable for those cases, in particular when changes are irreversible.[12]

In consequence, valuation usually focuses on the value of comparatively small (incremental or "marginal") *changes* in ecosystem services that result (or would result) from management decisions or from other human activities.[13] Some recent efforts have been made to derive the global (as opposed to incremental) value of ecosystems at a given time[14] and to simulate the value of ecosystem services in an integrated Earth system model.[15] However, the methodologies underlying these efforts, and the figures they produced, remain controversial;[16] moreover, as the Millennium Ecosystem Assessment notes, their usefulness for policy is limited, as it is rare for all ecosystem services to be completely lost and even then, such a complete loss would usually happen only over time.[17] For these reasons, and consistent with the

11 There has been considerable debate on how much biodiversity is necessary to keep the basic services of the planet intact. There is however a consensus that a more diverse ecosystem can provide services more reliably (Peterson et.al. 1998)

12 See Pagiola et al. 2005.

13 Changes in ecosystem services may also result from natural impacts that lead to a different state of the ecosystem. In such cases, management decisions may, for instance, include whether and how to mitigate these impacts on the ecosystem condition.

14 See Costanza et al. 1997.

15 See Boumans et al. 2002.

16 For instance, Dasgupta states that *"the value of an incremental change to the natural environment is meaningful because it assumes that humanity will survive the change to experience it. The reason (that) estimates of the total value (of the environment) should cause us to balk is that if environmental services were to cease, life would not exist."*. See Dasgupta 2000.

17 Millennium Ecosystem Assessment, volume one, chapter 2.3.3. This conclusion only applies to the limitations of valuation. Multi-disciplinary research will still be important to identify causal chains and interaction effects as well as the impacts of biodiversity loss.

approach chosen by the Millennium Ecosystem Assessment,[18] this report focuses on methods for assessing the value of *changes* in ecosystem services.

C. TOTAL ECONOMIC VALUE

The framework commonly used for describing the different *types* of economic value ascribed to natural resources is known as Total Economic Value (TEV). The framework comprises use values (direct, indirect and option value[19]) and non-use values. These types of value are summarized in figure 1.[20]

Direct use value is the value derived from direct use or interaction with environmental resources and services (e.g., timber, fuelwood, recreation are direct use values of a forest). They involve commercial, subsistence, leisure, or other activities associated with a resource.

Indirect use value relates to the indirect support and protection provided to economic activity and property by the ecosystem's natural functions. For example, carbon sequestration is a function of forest ecosystems whose value can be derived from the avoided costs of having to sequester by other means, or from avoiding the actual effects of warming. Similarly, the watershed protection function of a tropical forest may have indirect use value through controlling water quality and flood drainage that affect downstream agriculture, fishing, water supplies and other economic activities. While these functions have in principle long been recognized, precise field experimentation has often been lacking in order to show more precisely the relationships between ecosystem functions and the services generated.

Option value is a type of use value in that it relates to future use of the environment or biodiversity resources and functions. Option value arises because individuals may value the option to be able to use the natural resource some time in the future. For example, there may be an additional premium placed on preserving a forest system and its resources and functions for future use, particularly if prospects of future value are high and if current exploitation or conversion is irreversible.[21] The logic of the option motive is to maintain a diverse portfolio of resources as a means to reducing the risk of large fluctuations in value.[22] A more diverse ecosystem also tends to be considerably more resilient. This has been researched under the term "insurance value" as well.[23]

Quantification of option value is often complex. For instance, several attempts have been made to evaluate the expected benefits of bioprospecting of genetic resources of naturally occurring wild plants and organisms for pharmaceutical use. These attempts remain however controversial due to a number of open questions including: the role and extent of previous knowledge and its impact on probabilities of finding a resource of actual value; and the role and extent of potential replacement by human-made diversity.[24]

18 Ibid.
19 Option value is also sometimes classified as a non-use value.
20 It is important to not confound this concept with the attempts, explained above, to quantify the global (as opposed to incremental) value of ecosystem services worldwide.
21 See Conrad (1997) for an application to old-growth forest.
22 According to some ecological approaches on biodiversity, the value of a species could greatly vary, depending on the time of its loss. A species lost while others can still provide a given service and fill the gap might have a relatively low value, one lost later when it is the only one providing the service could cause the collapse of an ecosystem. Biodiversity becomes more valuable as it becomes scarce.
23 See Baumgaertner 2006.
24 See Simpson, Sedjo and Reid 1996, Rausser and Small 2000, Firn 2003.

TOTAL ECONOMIC VALUE AND VALUATION

TOTAL ECONOMIC VALUE (TEV)

	USE VALUE			NON-USE VALUE
TEV CATEGORIES	**Direct use value** Consumptive, noncon-sumptive	**Indirect use value**	**Option value**	**Existence value** Bequest value (for future generations)
EXAMPLES FOR BIODIVERSITY	• Hunting • Fishing • Timber harvesting • Harvesting of non-timber forest products • Harvesting of biomass • Recreation	**Watershed protection** (erosion control, local flood reduction, regulation of streamflows, storm protection) **Ecological processes** (fixing and cycling of nutrients, soil formation, circulation and cleansing of air and water, climate regulation, carbon fixing, global life support)	• Genetic resources • Old-growth forest (irreversibilities!)	Charismatic mega-fauna (whales, great apes, etc.)
COMMONLY USED VALUATION METHODS	Change in productivity, cost-based approaches, hedonic prices, travel cost, stated preference methods	Change in productivity, cost-based approaches, stated preference methods	Change in productivity, cost-based approaches, stated preference methods	Stated preference methods

The lack of knowledge is also reflected in the concept of a quasi-optional value,[25] referring to the value of having more time to learn about the extent[26] or possible uses of biodiversity, thereby making it possible for future decisions to be based on better information. Decisions should always take into account that the results of an interference with an ecosystem can never be fully predicted, making "ignorance a strong motivation for conservation."[27]

Non-use values such as existence value (sometimes also dubbed passive value) are derived neither from current direct or indirect use of the environment. For example, there are individuals who do not use the tropical forest but nevertheless wish to see it preserved because they simply derive utility from the ongoing existence of the ecosystem, or because they wish to conserve it for future generations (bequest value). A similar observation applies to some species, in particular charismatic mega-fauna such as whales or tigers. The concrete reasons why they derive utility may vary and may be based on, for instance, religious, spiritual, or ethical motives. In particular, a non-use motive may coincide with the recognition of an intrinsic right of existence. In this sense, valuation that is based on the concept of total economic value will also capture, at least to some extent, non-utilitarian values.[28]

Of all the value categories, existence or passive value is most complex in terms of quantification and its role in decision-making. Yet, it is a type of economic value that is significant in defining both national and global biodiversity management priorities.

25 See Weikard 2002.
26 According to biological estimates, as little as 10% of species are currently known.
27 Weikard 2002.
28 Millennium Ecosystem Assessment (2003), 133.

III. VALUATION METHODS

In the last decades, valuation methods have reached a considerable degree of sophistication. The last decades have also witnessed a gradually emerging consensus on the state-of-the-art of the range of valuation methods at hand, which is reflected by the fact that recent handbooks and manuals on the topic provide very similar overviews and assessments of the individual tools, with differences remaining essentially on the level of terminology and classifications.[29]

Valuation methods are increasingly applied not only in developed countries, but also in developing countries and countries with economies with transition. Rietbergen-McCracken and Abaza (2000) explain that:

> "[U]p to recently, there was considerable skepticism, particularly among international development organizations and developing country governments (as end users of the valuation results) about the possibilities of using valuation methods outside the relatively resource-rich and data-rich environments of developed countries. It was generally felt that developing countries and countries with economies in transition presented too many difficulties (including a scarcity of statistical information; the presence of price distortions or undeveloped markets; and in some cases largely illiterate communities) to allow valuation methods to produce meaningful results. However, over the last five to ten years a growing body of evidence has emerged to refute these claims."[30]

Rietbergen-McCracken and Abaza (2000) present a number of case studies of valuation studies undertaken in Africa, Asia, Latin America and Central and Eastern Europe, some of which also deal with biodiversity resources and functions, and the related ecosystem services. The IUCN guidelines for protected area managers on economic values of protected areas also provide summaries of a number of valuation studies in developing countries,[31] A survey on the use of contingent valuation studies in developing countries, some of which address biodiversity-related issues, was conducted by FAO in 2001.[32] Humavindu (2002) presents an analysis of valuation studies addressing nature-based tourism in Namibia.

The remainder of this sub-section is largely based on the review and assessment of valuation tools provided in chapter 2.3.3.1 of volume 1 of the Millennium Ecosystem Assessment. The reason for choosing this approach is that the report of the Millennium Ecosystem Assessment has already been extensively peer-reviewed by governments and experts.

Many methods for measuring the values of ecosystem services are found in the resource and environmental economics literature (Mäler and Wyzga 1976; Freeman 1979; Hufschmidt et al. 1983; Mitchell and Carson 1989; Pearce and Markandya 1989; Braden and Kolstad 1991; Hanemann 1992; Freeman 1993; Pearce 1993; Dixon et al. 1994; Johansson 1994; Pearce and Moran 1994; Barbier et al. 1995; Willis and Corkindale 1995; Smith 1996; Seroa da Motta 1998; Garrod and Willis 1999; Seroa da Motta 2001; Pearce et al. 2002; Turner et al. 2002; Pagiola et al. 2005).

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29 For recent handbooks and manuals, see e.g. IUCN (1998), OECD (2002), and Pagiola et al. 2005 (published by the World Bank). A manual that concentrates specifically on wetlands valuation has recently been published by the Bureau of the Ramsar Convention on Wetlands (de Groot et al. 2006).

30 Rietbergen-McCracken and Abaza (2000), 2.

31 IUCN (1998).

32 By discussing issues of relevance to successful implementation of this technique in these countries, the report can be used by FAO and its Member countries for guiding the work of practitioners who have a leading or technical contribution role in the design of CVM surveys. It is available under http://www.fao.org/es/ESA/en/pubs_wp01.htm .

Some techniques are based on actual observed behaviour data, including some methods that deduce values indirectly from behaviour in surrogate markets, which are hypothesized to have a direct relationship with the ecosystem service of interest. Other techniques are based on hypothetical rather than actual behaviour data, where people's responses to questions describing hypothetical markets or situations are used to infer value. These are generally known as "stated preference" techniques, in contrast to those based on behaviour, which are known as "revealed preference" techniques. Some techniques are broadly applicable, some are applicable to specific issues, and some are tailored to particular data sources. As in the case of private-market goods, a common feature of all methods of economic valuation of ecosystem services is that they are founded in the theoretical axioms and principles of welfare economics. These measures of change in well-being are reflected in people's willingness to pay or willingness to accept compensation for changes in their level of use of a particular service or bundle of services (Hanemann 1991; Shogren and Hayes 1997). These approaches have been used extensively in recent years, in a wide range of policy-relevant contexts.

Any one valuation method is unlikely to be able to cover *all* of the different types of value given in the concept of Total Economic Value.[33] Different techniques may also be required for the same biodiversity resource evaluated at different scales. For example, the range of services of a forest, the type of value of those services, and their actual value to a local community living at the fringe of the forest, may differ significantly from the types of value and the value that the national and/or international community may assign to different services of the same forest. The selection of the method or methods should therefore depend on which types of value, and on which levels, are deemed the most important or likely in a given situation.[34]

Valuation is a process involving several steps. First, the services being valued have to be identified. This includes understanding the nature of the services (bearing in mind that, under the Millennium Ecosystem Assessment understanding, services may also include goods) and their scale (being local, regional and/or global, on-site or off-site), and how they would change if the ecosystem changed; knowing who makes use of the services, in what way and for what purpose, and what alternatives they have; and establishing what trade-offs might exist between different kinds of services an ecosystem might provide. The bulk of the work involved in valuation actually concerns quantifying the biophysical relationships. In many cases, this requires tracing through and quantifying a chain of causality. Valuation in the narrow sense only enters in the second step in the process, in which the value of the impacts is estimated in monetary terms.

33 See, e.g., Nunes and van den Bergh (2001).

34 Accordingly, most of the valuation studies provided in the annex use several valuation methods, even while many of them deliberately restrict their attention to only one or several components of total economic value.

OVERVIEW OF VALUATION METHODS[35]

1. Changes in productivity

One widely used technique, thanks to its broad applicability and its flexibility in using a variety of data sources, is known as the change in productivity technique.[36] It consists of tracing through chains of causality so that the impact of changes in the condition of an ecosystem can be related to various measures of human well-being. Such impacts are often reflected in goods or services that contribute directly to human well-being (such as production of crops or of clean water), and as such are often relatively easily valued. The valuation step itself depends on the type of impact but is often straightforward.

The impact of hydrological changes on use of water for human consumption, for example, begins by tracing through chains of causality to estimate the changes in the quantity and quality of water available to consumers. This is itself often difficult. For instance, the relationship between tree cover and water productivity in a watershed is complex and often not well understood. Further scientific research into this relationship and the chains of causality will in such cases be a key precondition for valuation.

In the case of marketed goods, the actual valuation is relatively straightforward. For instance, the net value in reductions in irrigated crop production resulting from reduced water availability is easy to estimate, for example, as crops are often sold. (Even so, it is a very common error to use the reduction in the gross value of crop production rather than the net value. Using gross value omits the costs of production and so overestimates the impact.).

Where the impact is on a good or service that is not marketed or where observed prices are unreliable indicators of value, the valuation can become more complex. In the example above, it has to be noted that the prices charged to consumers for water consumption are typically not reliable measures of the value of the water to consumers, as they are often set administratively, with no regard for supply and demand (indeed, in most cases water fees do not even cover the cost of delivering the water to consumers, let alone the value of the water itself). The value of an additional unit of water can then be estimated in various ways, such as the cost of alternative sources of supply (cost-based measures are described later) or asking consumers directly how much they would be willing to pay for it (contingent valuation, described later). Note that it is very important to use the value of an additional unit of water, since some amount of water is, of course, vital for survival. Thus an additional unit of water will be very valuable when water is scarce, but much less so when water is plentiful. In this case, as in many others, averages can be misleading.

When the impact is on water quality rather than quantity, the impact on well-being might be reflected in increased morbidity or even mortality. Again, the process begins by tracing through chains of causality, for example by using dose-response functions that tie concentrations of pollutants to human health. Valuing the impact on health itself can then be done in a number of ways (see cost of illness and human capital, in the next section).

35 In accordance with the mandate set out in decision VII/18, this section focuses on the methodological status of the individual methods, but references to actual studies undertaken with these methods will also be given.
36 See studies III, IV, and X in the annex as specific examples.

In some cases, the impact is on relatively intangible aspects of well-being, such as aesthetic benefits or existence value. Starting in the 1960s, particular efforts have been made to develop techniques to value such impacts, including hedonic price, travel cost, and contingent valuation methods, and considerable progress has been made since then—see below for further discussion.

2. Cost of illness and human capital

The economic costs of an increase in morbidity due to increased pollution levels can be estimated using information on various costs associated with the increase: any loss of earnings resulting from illness; medical costs such as for doctors, hospital visits or stays, and medication; and other related out-of-pocket expenses. The estimates obtained in this manner are interpreted as lower-bound estimates of the presumed costs or benefits of actions that result in changes in the level of morbidity, since this method disregards the affected individuals' preference for health versus illness and restrictions on non-work activities. Also, the method assumes that individuals treat health as exogenous and does not recognize that individuals may undertake defensive actions (such as using special air or water filtration systems to reduce exposure to pollution) and incur costs to reduce health risks.

When this approach is extended to estimate the costs associated with pollution-related mortality (death), it is referred to as the human-capital approach. It is similar to the change-in-productivity approach in that it is based on a damage function relating pollution to productivity, except that in this case the loss in productivity is that of human beings, measured in terms of expected lifetime earnings. Because it reduces the value of life to the present value of an individual's future income stream, the human-capital approach is extremely controversial when applied to mortality. Many economists prefer, therefore, not to use this approach and to simply measure the changes in the number of deaths or in the probability of death (without monetary values), or measures such as disability-adjusted life years.

3. Cost-based approaches

The costs of replacing or restoring the services provided by the environmental resource can sometimes be relevant variables in decision-making.[37] For example, if ecosystem change reduces water filtration services, the cost of treating water to make it meet the required quality standards could be used. The major underlying assumptions of these approaches are that the nature and extent of physical damage expected is predictable (there is an accurate damage function available) and that the costs to replace or restore damaged assets can be estimated with a reasonable degree of accuracy. It is further assumed that the replacement or restoration costs do not to exceed the economic value of the service, bearing in mind that potential externalities generated by the replacement options should also be taken into consideration. These assumptions may not be valid in all cases. It simply may cost more to replace or restore a service than it was worth in the first place—for example, because there are few users or because their use of the service was in low-value activities.

Even while there is not necessarily any relationship between the replacement or restoration cost and the value of the service, cost-based approaches can provide useful guidance in a number of cases, in particular when the specific decision-making problem calls for a comparison of the costs resulting from all

37 See studies II, III, IV, V, and X in the annex.

different replacement or restoration options. For instance, in an often-quoted case, the New York City water authority avoided spending $6-8 billion on water purification plants by investing $1.5 billion for protection and restoration of the upstate watershed of the Catskills mountains.[38] Here, the decision-making problem was simply to minimize the cost of meeting an objective, by comparing the costs resulting from replacement and from restoration options. The priority given to the objective itself (a reliable supply of drinking water meeting certain quality standards) was unquestionable and, hence, not part of the decision-making problem.

4. Hedonic analysis

The prices paid for goods or services that have environmental attributes differ depending on those attributes. Thus, a house in a clean environment will sell for more than an otherwise identical house in a polluted neighbourhood. Hedonic price analysis compares the prices of similar goods to extract the implicit value (also dubbed "shadow price") that buyers place on the environmental attributes. This method assumes that markets are transparent and work reasonably well, and it would not be applicable where markets are distorted by policy or market failures. Moreover, this method requires a very large number of observations, so its applicability is limited.

5. Travel cost

The travel-cost method is an example of a technique that attempts to deduce value from observed behaviour in a surrogate market. It uses information on visitors' total expenditure to visit a site to derive their demand curve for the site's services. From this demand curve, the total benefit visitors obtain can be calculated. (It is important to note that the value of the site is not given by the total travel cost; this information is only used to derive the demand curve.)[39] This method was designed for and has been used extensively to value the benefits of site-seeing or of recreation at particular sites,[40] but it has limited utility in other settings.

6. Contingent valuation

Contingent valuation is an example of a stated preference technique.[41] It is carried out by asking consumers directly about their willingness-to-pay to obtain an environmental service.[42] A detailed description of the service involved is provided, along with details about how it will be provided. The actual valuation can be obtained in a number of ways, such as asking respondents to name a figure (classical CV), asking them whether they would pay a specific amount (dichotomous or polychotomous choice, paragraph 55) or having them choose from a number of options (choice modelling, paragraph 56).[43]

38 See Postel and Thompson 2005.
39 Technically, the total benefit is expressed as the area under the demand curve minus the costs—this is the sum of the consumer surplus and the producer surplus.
40 See studies I, VII, VIII, XII.
41 See studies II, VI, VII, VIII, IX, X.
42 Or, under some circumstances, their willingness-to-accept. See previous discussion.
43 Respondents do not necessarily have to provide a monetary figure. See the discussion on page 10 and study IV in the annex as an example.

Contingent valuation can, in principle, be used to value any environmental benefit simply by phrasing the question appropriately. Moreover, since it is not limited to deducing preferences from available data, it can be targeted quite accurately to ask about the specific changes in benefits that the change in ecosystem condition would cause. Because of the need to describe in detail the service being valued, interviews in contingent valuation surveys are often quite time-consuming. It is also very important to identify the relevant population, to ensure representativeness of the sample of respondents, and to have the questionnaire extensively pre-tested to avoid various sources of bias.

A potentially important limitation in terms of applying these methods to ecosystem services is that respondents cannot typically make informed choices if they have a limited understanding of the issue in question. Choosing the right approach for, and the adequate intensity of efforts in, improving the understanding of biological complexity of the sample group is a challenge for stated preference methods.

Contingent valuation methods have been the subject of severe criticism by some analysts, in particular because a number of biases can occur that would lead contingent valuation studies to not reflect true preferences:

- One major issue is that of so-called zero-bids, that is, respondents that state to have no willingness-to-pay at all. In some cases, such an occurrence can be explained by economic theory—the service in question is not valued by the respondent or his/her budget restrictions are too tight. However, zero-bids can also reflect protest—respondents who are not agreeing that they should pay for the service in question and who consider someone else responsible, for instance the government or the polluter. A zero-response may also be given when no trade-offs for the service are accepted at all (so-called lexicographic preferences). Finally, protest bids can also occur when the survey itself is rejected as a methodology, or payment vessels are not accepted.[44]
- Exaggerated willingness-to-pay statements are possible as well, for different reasons: (i) The phenomenon of "yea-saying" has been shown to occur sometimes—respondents will agree to a proposal or bid to please the interviewer or avoid further questions. (ii) The existence of a "warm glow" can also have an influence; respondents tend to feel good about giving, about being "good" or "nice", and will initially offer higher a willingness-to-pay than after thorough consideration. (iii) Strategic behaviour can also occur: participants will state unrealistic willingness-to-pay numbers in an attempt to influence the outcome of the study. (iv) Willingness-to-pay statements tend to also be elevated due to a lack of awareness of possible substitutes.
- Another source of bias can be through the interviewer giving information that is not fully neutral, or formulating questions to favour certain answers.

A "blue-ribbon" panel was organized by in the United States following controversy over the use of contingent valuation to value damages from the 1989 *Exxon Valdez* oil spill. The report of this so-called NOAA panel (Arrow et al. 1993) concluded that contingent valuation can provide useful and reliable information when used carefully, and it provided guidance thereon that can help to reduce or avoid many of the biases described above. This report is generally regarded as authoritative on appropriate use of the technique.

44 The payment vessel can be refused because another is considered superior (e.g. taxes vs. fees), or the responsible institution is not considered trustworthy.

The guidance of the panel includes *inter alia* the following requirements:

- The design of contingent valuation studies should be conservative, always rather allowing for an underestimate than an overestimate of willingness-to-pay.
- Because the concept of willingness-to-accept is a source of potential bias, willingness-to-pay should be preferred over willingness-to-accept.
- The valuation questions are to be asked as a vote on a referendum, not completely open.
- Sufficient information must be provided, however care is necessary in the use of pictures, including the pre-testing their effect on participants, and possibly making another choice.
- Participants should be made aware of substitutes for the good being evaluated.
- Sufficient time should pass after a negative impact on the ecosystem before a contingent valuation study is conducted in order to avoid answers out of a momentary disposition. Answers averaged over several points in time can avoid catching temporary changes in preferences.
- Respondents should be able to refuse an answer, with an attempt to be made of finding out the reasoning behind both refusals to answer and yes/no answers.
- A high quality survey would also include questions on socioeconomic data and respondents' general attitudes and perceptions of the issue at stake, with the influence of these variables on the willingness-to-pay being analyzed.
- Lastly, with all the above guidelines met, the questionnaire must still be easy enough to understand and not take an excessive amount of time to complete.

Dichotomous or polychotomous choice is a variant of Contingent Valuation where instead of open questions the respondents are asked whether they would pay a certain amount. Dichotomous choice allows only for "yes" and "no" answers, polychotomous choice provides more options such as "probably pay", "certainly pay" or "not sure". Questions can be single-bounded, where only one question is asked, or multiple-bounded, where follow-up questions with higher or lower amounts, depending on the initial reply, are asked. There are usually different versions of a questionnaire with different amounts being initially offered for choice. This technique makes answering easier for respondents, thereby reducing the chance of unrealistic statements. It does however bear the risk of starting point bias, that is, researchers influencing outcomes by choosing certain starting points.

7. Choice Modelling

Choice modelling (also referred to as contingent choice, choice experiments, conjoint analysis, or attribute-based stated choice method) is a newer approach to obtaining stated preferences.[45] It consists of asking respondents to choose their preferred option from a set of alternatives where the alternatives are defined by attributes (including the price or payment). The alternatives are designed so that the respondent's choice reveals the marginal rate of substitution between the attributes and the item that is trade off (e.g., money). These approaches are useful in cases in which the investigator is interested in the val-

45 See study VI for an example.

uation of the attributes of the situation or when the decision lends itself to respondents choosing from a set of alternatives described by attributes.

Choice modelling has several advantages: the control of the stimuli is in the experimenter's hand, as opposed to the low level of control generated by real market data; the control of the design yields greater statistical efficiency; the attribute range can be wider than found in market data; and the introduction or removal of products, services and attributes is easily accomplished (Louviere et al. 2000; Holmes and Adamowicz 2003; Bateman et al. 2004).[46] The method also minimizes some of the technical problems associated with contingent valuation, such as strategic behaviour of respondents. The disadvantages associated with the technique are that the responses are hypothetical and therefore suffer from problems of hypothetical bias (similar to contingent valuation) and that the choices can be quite complex when there are many attributes and alternatives. The econometric analysis of the data generated by choice modelling is also fairly complex.

8. Benefits transfer

A final category of approach is known as benefits transfer. This is not a methodology *per se* but rather refers to the use of estimates obtained (by whatever method) in one context to estimate values in a different context. For example, an estimate of the benefit obtained by tourists viewing wildlife in one park might be used to estimate the benefit obtained from viewing wildlife in a different park. Alternatively, the relationship used to estimate the benefits in one case might be applied in another, by using adjusted data from this case in conjunction with some data from the site of interest ("benefit function transfer"). For example, a relationship that estimates tourist benefits in one park, based in part on their attributes such as income or national origin, could be used in another park, but with data on income and national origin of that park's visitors.[47]

Benefits transfer has been the subject of considerable controversy in the economics literature, as it has often been used inappropriately.[48] According to the Millennium Ecosystem Assessment, a consensus seems to be emerging that benefit transfer can provide valid and reliable estimates under certain conditions. These conditions include the requirement that the commodity or service being valued be very similar at the site where the estimates were made and the site where they are applied and that the populations affected have similar characteristics.[49] Of course, the original estimates being transferred must themselves be reliable in order for any attempt at transfer to be meaningful.

As the conditions at the two sites are unlikely to be perfectly identical, some transfer error is to be expected. This feature, however, does not speak *as such* against the application of benefits transfer in real-world decision-making. This is because estimates based on benefits transfer can be generated with considerably less time and resources than primary studies. In a world of scarce resources and typically very costly primary studies, decision makers may be willing to trade quick and cheap numbers against a certain loss in accuracy, provided that minimum quality standards are met. They may even be more

46 Conjoint analysis to value ecosystem services in different rural areas has been used in Colombia in a project by the Alexander von Humboldt Institute in cooperation with the University of Massachusetts. See Colombia 2002.
47 See in particular study XI in the annex.
48 See Brouwer 2000; Christie et al. 2004, 40, for further discussion.
49 Up to a limit, differences in the population's characteristics can be addressed by using benefits functions transfer.

ready to do so when the relevant alternative, under given resource constraints, is simply to have no estimate at all. Moreover, benefits transfer may be attractive when decision makers request, as is frequently the case, quick (but not necessarily final) answers from administrators—it may hence play a role within rapid assessment methodologies.[50]

9. Summary assessment of valuation methods

Each of the approaches reviewed above has seen extensive use in recent years, and considerable literature exists on their application. These techniques can and have been applied to a very wide range of issues (Rietbergen-McCracken and Abaza 2001), including the benefits of ecosystems such as forests (Bishop 1999; Kumari 1995; Pearce et al. 2002; Hanley et al. 2002, Merlo and Croitoru 2005), wetlands (Barbier et al. 1997; Heimlich et al. 1998; de Groot et al. 2006), watersheds (Aylward 2004; Kaiser and Roumasset 2002). Other studies have focused on the value of particular ecosystems services such as water (Young and Haveman 1985), non-timber forest benefits (Lampietti and Dixon 1995; Bishop 1998), recreation (Bockstael et al. 1991; Mantua at al. 2001; Herriges and Kling 1999; Humavindu 2002), landscape (Garrod and Willis 1992; Powe et al. 1995), biodiversity for medicinal or industrial uses (Simpson et al. 1994; Barbier and Aylward 1996), natural crop pollination and cultural benefits (Pagiola 1996; Navrud and Ready 2002). Many valuation studies are catalogued in the Environmental Valuation Reference Inventory Web site maintained by Environment Canada (EVRI)[51] or the ENVALUE environmental valuation database developed by the New South Wales Environmental Protection Agency of Australia.[52]

It appears that, when applied carefully and according to best practice, valuation tools can generally provide useful and reliable information on the changes in the value of non-marketed ecosystem services that result (or would result) from management decisions or from other human activities. Data requirements may be quite demanding for a number of tools, as are the preconditions in terms of technical expertise. Moreover, conducting primary valuation studies is typically time-consuming and costly.

According to the Millennium Ecosystem Assessment, measures based on observed behaviour are generally preferred to measures based on hypothetical behaviour, and more direct measures are preferred to indirect measures. However, it is also pointed out that the choice of valuation technique in any given instance will be dictated by the characteristics of the case, including its scope, and by data availability.

Several techniques have been specifically developed to cater to the characteristics of particular problems. The travel cost method, for example, was specifically developed to measure the utility derived by visitors to sites such as protected areas, and could also be applied to similar areas of interest, but is of limited applicability outside that particular case. The change in productivity approach, on the other hand, is applicable to a wide range of issues.

Contingent valuation is potentially applicable to any issue, simply by phrasing the questions appropriately and as such has become very widely used—probably excessively so, as it is easy to misapply and, being based on hypothetical behaviour, is inherently less reliable than measures based on observed

50 Christie et al (2004) note in this connection that: “Finding acceptable benefits transfer methods is essential to the wider use of environmental valuation in policy. However, the standards of accuracy required in academic work may exceed those viewed as tolerable by policy-makers. (…) The key question is: how close is close enough for policy purposes?”

51 http://www.evri.ca .

52 http://epa.nsw.gov.au/envalue/ .

behaviour. For instance, if the focus is on the quantification of indirect use values, the application of other valuation tools would often seem to be preferable. For some types of value, however, stated preference methods may be the only alternative. Thus, existence value can only be measured by stated preference techniques. Guidance on the appropriate use of the technique exists and should be followed closely.

Benefits transfer has often been used inappropriately but can provide valid and reliable estimates under certain conditions. Given the cost of undertaking primary valuation studies, benefits transfer when used cautiously is likely to be an increasingly appealing way for extending the use of valuation, including in developing countries.

IV. VALUATION AND DECISION-MAKING

As said earlier, undertaking valuation has the potential of improving public decision-making on projects or regulations as well as, under specific circumstances, of improving legal decision-making. In this connection, the synthesis report of the Millennium Ecosystem Assessment also notes that:

> "[M]ost resource management and investment decisions are strongly influenced by considerations of the monetary costs and benefits of alternative policy choices. Decisions can be improved if they are informed by the total economic value of alternative management options and involve deliberative mechanisms that bring to bear non-economic considerations as well."

Existing methods to support decision-making use valuation information to a greater or lesser extent. Economic frameworks such as cost-benefit analysis (CBA) and cost-effectiveness analysis (CEA) involve explicit monetary valuation. An important advantage of the valuation tools reviewed in the last section is that they provide numbers in a common (monetary) metric, which can thus easily be incorporated into these standard appraisal methods. In contrast, multi criteria analysis (MCA) typically avoid using a monetary unit of account. Other non-economic approaches to prioritization include deliberative processes, scorecard approaches, expert judgment and satisficing.

All of these approaches are but *tools* to *support* decision-making. All of them have specific advantages and limitations, and it cannot be claimed that one tool is generally superior, or that it should be used as an *exclusive* tool in decision-making. For instance, with regard to cost-benefit-analysis, it has to be acknowledged that economic efficiency is seldom the sole criterion for public investment decisions. The distributional impacts of decisions are often also important.[53] While cost-benefit-analysis can be helpful in clarifying distributional impacts, it does not deliver recommendations with regard to preferable decisions from a distributional perspective.[54] It will be shown in the subsequent paragraphs that the different methods may be used in a complementary manner in order to support decision-making.

A. ECONOMIC FRAMEWORKS

1. Cost-benefit analysis and cost-effectiveness analysis

Cost-benefit analysis compares monetary costs and benefits in commensurate terms. This comparison is sometimes expressed as a cost-benefit ratio, with benefits as the numerator and costs as the denominator. Alternative options can then be ranked in accordance with their cost-benefit-ratio. Depending on the specific activities under investigation, the value associated with ecosystem services will be included as a cost or as a benefit. For instance, if the cost-benefit-ratios of different conservation projects were compared, the value of improved ecosystem services would be included as benefits of the

53 For instance, many direct use values in developing countries arise in the context of subsistence activities that are often crucially important to rural populations. A range of studies has concentrated on the links between poverty alleviation and the sustainable exploitation of naturally occurring products. See e.g. Cavendish 1999 and 2003.

54 For instance, while the benefits of conservation are often widespread—they may occur at national or even international scales, such as in the case of carbon sequestration or the existence value associated with charismatic mega-fauna, the costs associated with conservation, in terms of foregone exploitation of resources or of use of land, are often borne by local populations. On the other hand, local ecosystem services—those that benefit local communities—are also severely undervalued in many cases—see studies III and IV for examples. By suitably disaggregating the numbers obtained, valuation studies and cost-benefit-analysis can help to quantify the shares of local vs. national vs. regional or international benefits and costs under different options.

individual projects. If, however, different development projects were considered, such as for instance different options to invest into public infrastructure with negative impacts on biodiversity, the value of the associated loss of ecosystem services would be included as a cost to the individual option.[55] It is noteworthy that it may not be necessary to explore the full range of services of a given ecosystem in order to have an influence on the policy outcome. This will be the case whenever the benefits associated with the most important ecosystem services are already high enough to tip the balance—within cost-benefit-analysis—against a specific development option.

As costs and benefits typically occur at different points in time, some way must be found to collapse the recognized cost and benefit flows to a commensurate basis. This conventional economic process is known as time discounting, and the outcome of this process is called the "present value" of costs and benefits. A crucial variable in the calculation of present values is the choice of a discount rate; i.e. the value that is used to collapse future values to their present equivalents. Any positive rate of discount is tantamount to saying that the future (costs and benefits) are worth less in relative terms than the present, that is, costs and benefits that are realized immediately.

For conventional investment purposes, the rate of discount is simply the relevant market interest rate. But when it comes to choosing the appropriate rate for making a judgement on government projects or policies with important social and environmental impacts, important ethical and philosophical issues arise that relate to the status of present versus future (possibly unknown) preferences. Aspects of intergenerational justice play an important rule in that discussion, since the value of future benefits is only smaller to the current generation, whereas biodiversity will be of importance for many generations to come. While most contributors would seem to agree that the rate of discount should be positive, the correct number is the subject of much debate.

Discount rates used in public decision-making tend to vary between 3 and 15 per cent across different countries. The choice of discount rate is in the first instance be guided by the rate that is used by the public sector for appraising its other investments, which implies that biodiversity-related "investments" would be treated like all other investments. In many cases, however, lower rates are used. One important reason why not to accept a standard discount rate for biodiversity has been advanced in the shape of the so-called Krutilla-Fisher method.[56] Even if future preferences for biodiversity are uncertain, current trends mean that the future of many biodiversity components and resources is looking bleak, implying that they will be increasingly scarce, or more valuable, in the future. As future generations will place a higher value on scarcer resources than current generations, this reasoning then gives rise to a positive premium on the future, which offsets the discounting process described above.

Cost-effectiveness analysis (CEA) leaves the numerator in qualitative terms and simply compares the different costs of attaining some objective stated in the numerator. Different options *that deliver the same objective* are then compared and prioritized based on their cost-effectiveness-ratio.[57] CEA, therefore, does not ask nor attempt to answer the question of whether the goal of the policy is justified, in the

55 This procedure would allow to capture what economists call the "opportunity cost" of the individual development project, that is, the cost in terms of the most valuable opportunity foregone and the benefits that could be received from that opportunity. The consideration of opportunity costs is one of the key differences between the concepts of economic cost and accounting cost.
56 See Pearce and Turner 1990, Krutilla and Fisher 1975, Hanley and Craig 1991.
57 Alternatively, CEA may assume a fixed budget and seeks the alternative that will result in the maximum effect on a specific target variable.

sense that the social benefits expected from this goal exceed the costs necessary to reach the goal. In fact none of the options may be economically efficient, in the sense of monetary economic costs outweighing economic benefits. Hence, CEA is appropriate whenever there are good reasons to believe that the benefits of meeting the objective outweigh the costs, and the priority given to meet the objective is therefore not under doubt.[58] In other cases, however, CEA may only be helping to select the least worst option among a list of (potentially) inefficient options. Even in those cases, CEA is sometimes used as a second-best option when a full-blown CBA would be desirable, but many benefits cannot easily be monetized.

Both CBA and CEA are common governmental appraisal methods in OECD countries and among international organizations. While the methods were originally developed for appraising basic infrastructure, many government guidance documents now include advice on the inclusion of environmental and social costs and benefits.[59]

2. National income accounts

While CBA and CEA are decision-making tools relevant to projects and regulations, national income accounts are a key indicator framework for setting priorities in domestic macroeconomic policies. National income accounts are a long-standing economic convention by which economic performance are measured. In essence, the accounts measure national output from all sources (known as gross domestic product), and then deduct a measure of depreciation, which is the amount of (typically) man made capital that is used up in production. The result is a figure that depicts, in economic terms, how well off a country is year on year. While conventional accounts already include many biological products (e.g. production of timber and fish), in the last two decades there have been numerous attempts, at national and international levels, to include environmental externalities and, more importantly, some measure of environmental depreciation to reflect the environmental losses that occur as a result of economic activities. The United Nations Statistics Division, together with other organisations, has developed the System of Economic and Environment Accounting (SEEA). It was introduced in 1993 and revised in 2003, and consists of a satellite system to the UN System of National Accounts (SNA), in which changes in important natural assets are accounted for in physical terms.[60]

Due to the problems involved in assessing values in a comprehensive manner, most work in this area focus on those types of value that can be measured comparatively easily. It mainly includes direct uses values that are traded on markets, and opportunity costs for protected areas, sometimes also the impact of pollution. Some methods focus on the use of natural resources, both renewable and non-renewable as an indicator for the use of nature. For instance, recent work of the World Bank on an adjusted GDP and adjusted measurements of national capital stocks adopted the concept of genuine savings or adjusted net savings, which measure the true rate of savings in an economy after taking into account investments in human capital, depletion of natural resources and damage caused by pollution.[61] The work shows that several countries that perform well on conventional grounds were actually performing less well once the new measure of depreciation was included. Under a more comprehensive valuation exercise, including most indirect and non-use values, the number of countries with this sobering feature

58 The Catskills case, discussed above, provides an example.
59 See for example chapter 4 of the United Kingdom Treasury Green Book on public appraisal. http://greenbook.treasury.gov.uk/
60 See http://unstats.un.org. A United Nations Committee on Environmental-Economic Accounting was established to promote and implement this work.

would arguably be higher. Identification of this environmental drag on economic growth can serve as a basis for prioritizing national environmental policies and a focus on mitigation or reversal of environmentally damaging activities.

While valuation is central to the exercise of environmental adjustment of national accounts, many theoretical and methodological challenges remain with regard to an adequate incorporation of biodiversity values in conventional macro economic indicators of growth.[62] For instance, many of the valuation tools at hand are simply too costly and demanding to apply them on a scale that would be needed for a *comprehensive* valuation of the annual changes in domestic biodiversity resources.[63] Nevertheless, national income accounts remain an important vehicle into which more information about biodiversity loss must be directed. Further research directed at the development of a biodiversity adjustment is an important means to have biodiversity losses more reflected in macroeconomic discourse.

B. NON-ECONOMIC FRAMEWORKS

The economics approaches mentioned so far are all potentially informed by the tools for the valuation of biodiversity resources presented in the previous section. The following approaches are more qualitative in nature but may occasionally use valuation information in the decision process.

1. Multi criteria analysis

Multi-criteria analysis (MCA) is in fact a family of methods that use different scoring approaches to weigh the different attributes of a decision. They are used to structure a policy problem in terms of possible policy alternatives and to assess each alternative under various criteria. Most of the variants of MCA are structured approaches used to determine overall preferences among alternative policy measures, where each policy measure may pursue several objectives. Participants in the analysis are typically given the criteria that define different options and are asked to score or weigh these criteria using some pre determined points system.

Multi-criteria analysis is mainly applicable to cases where a single-criterion approach is insufficient. Instead, an MCA may accommodate a range of social, environmental, technical, economic, and financial criteria. MCA is therefore applicable especially where significant environmental and social impacts are present, which cannot (easily) be expressed in monetary terms.[64] MCA are often integrated with deliberative and participatory approaches and are said to facilitate such input to a larger degree than the monetary assessment tools CBA and CEA.[65]

61 The work is based on Pearce & Atkinson.
62 See for instance Nordhaus and Kekkelenberg (eds) 1999 for further discussion.
63 Such problems are not alien even to conventional National Accounting. For instance, many of the issues involved in including environment values at the macro level are not uncommon to problems involved in measuring the cost of living. See Nordhaus and Kekkelenberg (1999).
64 Biodiversity indicator frameworks may play a key role in assessing the impacts of the project or policy under consideration. See, for further information, the guidance, lessons learned and list of indicators provided in the note by the Executive Secretary on monitoring and indicators prepared for the ninth meeting of SBSTTA (UNEP/CBD/SBSTTA/9/10) (paragraph 8 of decision VII/8, on national-level monitoring programmes and indicators, refers to this document).
65 See Nichols et al. 2000 for further discussion.

There are very few applications of MCA in developing countries. MCA is often difficult to use and understand for lay people. Most variants require an expert to explain how the method works, and to help users to define options, criteria and weights, as well as to choose the appropriate aggregation procedure. The method also makes no claim to be searching for economically efficient outcomes. Like CEA, all options under consideration may be inefficient.

CBA and MCA are not mutually exclusive. CBA can be used to define a set of efficient options, that is, options where net benefits are positive (that is, gross benefits are greater than costs). Options with net economic benefits of similar magnitude could be further assessed by MCA so as to identify the various non-economic trade-offs associated with the alternative courses of action.

2. Deliberative and participatory approaches

Deliberative processes (sometimes also referred to as "deliberative and inclusionary processes" or "DIPs") include participatory appraisal, focus groups, Delphi approach, consensus conferences and citizen's juries. These methods are aimed at creating better informed decisions that are owned by and have the broad consent of all relevant actors and stakeholders. They therefore contrast to the more "technocratic" approaches such as cost-benefit or cost-effectiveness analysis or even MCA. DIPs seek to build a process of defining and redefining interests that stakeholders introduce as the collective experience of participation evolves. As participants become more empowered, i.e. more respected and more self-confident, so it is assumed they may become more ready to adjust, to listen, to learn, and to accommodate to a greater consensus.[66]

In many countries, the benefits emanating from some ecosystem services are well known to local and indigenous communities—it is captured by their *traditional knowledge*. As long as these communities are adequately included in economic valuation exercises (for instance, by ensuring that they are adequately represented in the population sample for a stated preference study), the value *they* put on these ecosystem services would be captured by economic valuation.

However, traditional knowledge of ecosystem services is often not adequately received by the wider public. Here, deliberative and participatory approaches may play an important role in promoting the wider recognition of this knowledge. It may also contribute, with the approval and involvement of these communities, to its wider application including within economic valuation studies. For instance, it was explained above that a limitation of stated preference techniques is that respondents cannot typically make informed choices if they have a limited understanding of the issue in question. Deliberative and participatory approaches, by disseminating pertinent knowledge, may play an important role in broadening the understanding on the issue of all stakeholders.[67]

The adoption of such methods varies across countries, with some having formal processes for undertaking participation in the formulation of contentious area of public policy. The use of economic information in these methods is entirely at the group's discretion. Hence, valuation data may or may not consistently inform the outcome of such processes, and they cannot guarantee that outcomes are an efficient

66 De Groot et al. 2006, p. 19 describes in detail the process of identifying and involving relevant stakeholders.
67 See studies IV and IX as examples of such application of deliberative approaches.

use of public resources. Moreover, in many countries, the relative weight that the outcome of these processes is given in final decisions is unclear.

3. Satisficing

A satisficing approach can be described as an assessment procedure to obtain an outcome that is good enough, rather than seeking the best solution. The approach can thus be contrasted with an optimizing approach that seeks to identify the "best" solution, as is the case, for example, with cost-benefit analysis or multi-criteria analysis. For the implementation of a satisficing approach, one or more criteria need to be identified that the measure is expected to fulfil. The subsequent analysis can then either investigate all possible measures to achieve this objective(s), and list the successful options without ranking them. Alternatively, the analysis may also be terminated once the first option has been identified that fulfils the requirement(s).

In decision theory, the term satisficing is also used to refer to an optimization process where *all* costs, including the cost of the optimisation calculations and the cost of getting information for use in those calculations, are considered. This takes account of the fact that, in some cases, the costs of gathering and processing information may not be justified by the subsequent improvements in decision-making that can be achieved through the improved information. This is likely to be the case in decision situations with a low level of complexity, where only few well-defined options are available, where the targets are clearly specified and where little or no trade-offs between targets are necessary.

One difficulty associated with such an approach is that the added value of better information for the decision-making process may only be apparent if this information is available: if it is not available, it may be hard to assess in what way better information might have changed the results of the decision, and what impact this would have had.

4. Liability and redress

In some countries, the legal framework for liability and redress priorities has been an important driver for the analysis and refinement of valuation methods. For instance, in the United States, the ability to use valuation information as the basis for legal redress has been a significant impetus for considering the value of damaged biological resources. High damage costs, derived including through non-market valuation have given plaintiffs a large incentive to demonstrate the monetary value of any damaged resources. As a result, valuation methods, and contingent valuation in particular, has come under considerable scrutiny in high profile legal cases such as the *Exxon Valdez* oil spill, with guidelines having been developed for the appropriate use of stated preference techniques (NOAA 1994)[68] Under the natural resource damage assessment (NRDA) regulations of the National Oceanic and Atmospheric Administration (NOAA), valuation (the so-called value-to-value approach) is applicable when the injured and restored resources and services are not of the same type, quality, and value, and is used to calculate the value of gains from the proposed restoration actions and the value of the interim losses.[69]

68 Theses guidelines, developed by a panel of eminent economists, are now generally accepted as authoritative for a reliable survey methodology in contingent valuation. See under III 6 above for further discussion.

69 See Penn 2005 for a brief summary. Interim losses are losses of welfare incurred to persons during the time that it takes for an ecosystem to recover or to be restored. If an ecosystem cannot be fully restored, or takes a long time to do so (e.g. trees growing back), interim losses can be large.

The European Union also explored the legal basis for using of non market values as evidence. Under Directive 2004/35/CE on environmental liability with regard to the prevention and remedying of environmental damage, it is within the discretion of the competent authority to use economic valuation to determine the extent of the necessary complementary and compensatory remedial measures, if it is not possible to use the so-called resource-to-resource or service-to-service equivalence approaches.[70] Compensatory remediation shall be undertaken to compensate for the interim loss of natural resources and services pending recovery. This compensation consists of additional improvements to protected natural habitats and species or water at either the damaged site or at an alternative site. It does not consist of financial compensation to members of the public. If valuation of the lost resources and/or services is practicable, but valuation of the replacement natural resources and/or services cannot be performed within a reasonable time-frame or at a reasonable cost, then the competent authority may choose remedial measures whose cost is equivalent to the estimated monetary value of the lost natural resources and/or services.

In many other countries, however, weak legal systems, poorly defined and enforced property rights over damaged resources, and/or the fact that many damaged resources are governed by customary law or practices that are not necessarily recognized by legal systems in a national context, mean that (formal) legal drivers for the application of valuation tools are currently weak to non-existent.

C. CONCLUSIONS

The use of formal appraisal methods and the nature of decision-making processes generally vary across countries. Even when formally documented procedures are in place it is impossible to generalize how and when different methods are most appropriate. In general, methods such as cost-benefit analysis seem to be less controversial, and are commonly applied, when financial costs and benefits are relatively clear to identify and when for instance social impacts are comparatively small. There seems to be a need to include decision making tools that are more consensual and participation oriented, in particular when external costs have significant social consequences, when they are captured by traditional knowledge that is not widely available, and/or when the local socio cultural systems pose a serious limitation to valuation based solely on economic terms. The combined utilization of different decision making tools may be useful.[71]

Mirroring the research progress made in developing reliable tools and methodologies, valuation studies in many countries play an increasing role in contemporary environmental policies, as they provide additional knowledge to support better decision-making. However, the integration of valuation information into decision-making frameworks still seems to be not satisfactory in many countries.

Conducting primary valuation studies is time-consuming and costly. Given the limited budget and manpower in many administrations, the need to conduct or manage primary research can pose a strain on the available resources. Capacity, both in conducting valuation studies and in overseeing their prepa-

70 Under these approaches, actions that provide natural resources and/or services of the same type, quality and quantity as those damaged shall be taken, or alternative natural resources and/or services shall be provided. See Annex II, paragraphs 1.2.2 and 1.2.3, of the Directive. See EC (2001) for background information. In Canada, a recent Supreme Court decision confirmed the acceptance of valuation approach to support damage valuation.

71 See studies studies IV and IX as examples.

ration and ensuring their quality, is often limited. Problems are exacerbated when the rationale for valuation is poorly conveyed to higher-level administrators. In many cases, the people that matter can be left with an impression that new research will not produce added value for the quality of decision-making. More commonly, poorly conducted studies,[72] with limited follow-up, can leave officials with an impression that valuation studies can only tell them what they already know. Resources that flow into the studies become harder to justify.

Hence, it is important to apply and interpret valuation results in their appropriate context and to be aware of the pitfalls involved. However, this applies to most methods and techniques, whether in economics or in any other field. Many basic criticisms levelled at valuation can be avoided when best practice is followed while conducting valuation studies; for example, a contingent valuation study can well be integrated with and extended into a public participation exercise. The main question is rather—given their high costs and the expertise required—how their use can be targeted at those cases where valuation studies actually provide an added value in terms of improved decision-making.

Resistance to the use of valuation in OECD countries has in recent years been addressed by attempts to produce both valuation guides and protocols as well as standard environmental values for use in benefits transfer. These efforts have revealed to be fruitful in terms of increasing the credibility and acceptability of valuation methods. More importantly, these resources have also simplified and reduced the cost of undertaking policy appraisal.

72 For instance, a problem frequently identified in the literature is high values derived though contingent valuation studies. For many observers excessive stated preferences defy intuition and apparently discredit the method. See section II B above for further discussion.

V. STRENGTHEN INTERNATIONAL COLLABORATIVE PARTNERSHIPS FOR ASSESSING BIODIVERSITY VALUES

Valuation is beginning to play a significant role in biodiversity management decisions in OECD countries. Many Governments espouse its use, with the predominant framework being cost-benefit analysis, even if they acknowledge the technical difficulties of consistent implementation of valuation in decision-making. While it would be premature to suggest that biodiversity values are always consistently considered, the important thing to note about the experience is that there is a formally documented approach that should be followed in determining resource allocations and in setting priorities.[73]

International organizations such as the OECD, the European Community, the World Bank and the Global Environment Facility (GEF) have all advocated greater use of valuation in policy making and project design. Other United Nations organizations, such as the United Nations Development Programme (UNDP), the United Nations Environment Programme (UNEP) and the Food and Agriculture Organization of the United Nations (FAO) have, at various times, either sponsored meetings on the topic or undertaken projects, which have a strong biodiversity valuation component. Several governments have also facilitated greater use through the sponsorship of meetings and information databases sources such as EVRI or ENVALUE, mentioned above.[74]

Similarly a number of non-governmental organizations, such as IUCN, WWF and Conservation International, have continued to sponsor research and wider application and dissemination about biodiversity valuation, and its role in creating incentives for conservation and sustainable management of biodiversity.

A combination of poor institutional capacity and a lack of trained staff can generally be identified as the main barriers to further promotion of valuation as a biodiversity management tool, in particular in developing countries and countries with economies in transition. Overall, valuation can normally be advanced in most countries by the development of high profile studies that help to raise the issue of biodiversity in national debates.[75] Many countries have reached this stage, but many others have not. This critical phase requires international collaboration and enhancement of domestic capacity.

A. INSTITUTIONAL CAPACITY

Poor institutional capacity is often an important impediment for consistent policy and regulatory appraisal. However, even where staff and infrastructure are relatively adequate, institutional weakness manifests in poorly defined lines of responsibility and the absence of clearly defined governmental practices for appraising basic policy changes such as projects and regulations.

73 In the United Kingdom for example, there is clear guidance on the importance of considering non-market values in central government project and regulatory appraisal. This guidance in nominally the responsibility of the Treasury (Ministry of Finance), which advocates good appraisal practice across a range of government ministries, including environment and transport. This model of appraisal practice is mirrored in several other countries.

74 Other databases include the Ecosystem Services Database (ESD) developed by the Gund Institute for Ecological Economics, University of Vermont; the Valuation Study Database for Environmental Change in Sweden (VALUEBASE SWE; see http://www.beijer.kva.se/valuebase.htm), and the Review of Externality Data database developed by the European Commission under the Energy, Environment and Sustainable Development Program of DG Research (http://www.red externalities.net/). Lantz and Slaney (2005) provide a recent evaluation of different valuation databases.

75 Typically one finds that an exercise such as a national accounts adjustment or one study of an endemic or charismatic species is sufficient to kick-start a national debate on the topic.

These institutional weaknesses can be summarized as a checklist, which can in turn provide a basis for the identification of needs:

(a) Does a single ministry or agency hold a clearly defined remit for biodiversity management?

(b) Is there a formally documented procedure for conducting environmental impact assessments of new projects and regulations?

(c) Is there a formally documented economics appraisal process for: (i) new projects; and (ii) new regulations?

(d) Do the project or regulatory appraisal procedures include quantitative as opposed to qualitative assessments of costs and benefits?

(e) What role does cost-benefit analysis play in appraisal?

(f) Do formal guidelines for cost-benefit analysis exist?

(g) Do formal guidelines include guidance on how to deal with non-market costs and benefits and biodiversity in particular?

(h) What is the legal status of the findings of government appraisals?

(i) Who conducts any appraisal and how is the process audited for quality control?

B. CAPACITY BUILDING AND TRAINING

The answers to many of the preceding questions are likely to be qualified by the issue of adequate capacity building and training. At the governmental level, capacity needs to be enhanced, by appropriate training, for conducting the actual valuation studies, for improved oversight and auditing for quality control, as well as for putting valuation results to good use in governmental decision-making by an effective and credible follow-up. Moreover, training could also include staff of relevant non-governmental organizations. In accordance with national needs and priorities, institutional capacity could be enhanced, for instance by establishing or strengthening of specialized agencies or agency units.

As regard the conduct of the actual studies, two levels of training seem to be required: first, basic courses are needed to provide non-economists with sufficient insights into the logic of valuation and environment economics. To meet this objective, easily readable manuals could also be developed and disseminated. Courses may also be needed to provide economists with basic scientific background on the linkages between biodiversity resources and functions and ecosystem services, with a view to raise their awareness of the need for inter-disciplinary cooperation. Second, more specialized training is needed for those supervising the conduct of valuation studies and steering project implementation, which is likely to be best undertaken by economists equipped with prior knowledge in microeconomics. Well-planned modules can normally be sufficient to impart the basics of environmental valuation to trained economists. This activity could also include the training of trainers.

1. International cooperation in enhancing domestic capacities

Most expertise in valuation is arguably located in several OECD countries that have established research institutions specifically in the area, and it appears to be important to tap into this expertise as a basis for sponsoring training arrangements such as regional workshops on biodiversity valuation.

In non-OECD countries, notable centres of expertise are, for instance, the Environment and Economics Program for South East Asia (EEPSEA)[76] and the Forum for Economics and the Environment located in South Africa.[77] EEPSEA offers courses that are predominantly for post-Masters level ability in economics. The South African initiative coordinates exchange of information and training between the countries of Southern Africa. In addition, in many developed countries, many university departments offer exchange opportunities that are normally supported by their own national development ministries (e.g., the Swedish International Development Agency (SIDA), Danish Ministry for Foreign Affairs). Short-term courses are offered by other agencies including the World Bank, which offers a course in environmental economics and development policy.

Another means of extending training is for bilateral arrangements between agencies for temporary secondment. For instance, the Overseas Development Institute (ODI) in the United Kingdom has been running such a fellowship scheme for several decades. The scheme sends young postgraduate economists to work in the public sectors of developing countries in Africa, the Caribbean and the Pacific on two-year contracts. It has worked in over 30 countries concentrating on those most in need of trained staff. Currently, 20 developing country Governments and three regional bodies are partners in the scheme.

The demand-led nature of the scheme means that it is an attractive way for Governments to build capacity in their public sectors and improve the execution of economic policy. Its excellent reputation and unique form of technical assistance means that it is held in high regard by the development community. The costs of the scheme are shared between the recipient Government and ODI. ODI finances the scheme primarily under grants provided by the Department for International Development (DFID) in the United Kingdom, the Commonwealth and AusAID. In recent years, the scheme has been picking a number of graduates in environmental economics who have gone to work in environment related agencies.

2. Web-based resources

Possibly the most cost-effective partnership arrangements can de developed using web access. The World Bank offers a range of e-learning resources. Moreover, several sites provide good overviews of environmental valuation. A simple and accessible site is provided by Dennis King, of the University of Maryland, and Marisa Mazzotta, of the University of Rhode Island.[78] This site sets out all the relevant issues in relation to valuation and contains some practical demonstrations of how to collect relevant data. The site provides an excellent introduction, but those who follow it probably need to be faced with more complex case studies to gain hands-on experience. In this regard, other web-based learning and training resources are also available, with varying levels of theory and applied examples. For instance:

(a) A UNEP sponsored training guide on the valuation of biological diversity for national biodiversity action plans and strategies is available at the IUCN biodiversity economics site;[79]

76 http://www.idrc.ca/en/ev-7890-201-1-DO_TOPIC.html
77 http://www.econ4env.co.za/.
78 http://www.ecosystemvaluation.org/
79 http://www.biodiversityeconomics.org/valuation/topics-612-00.htm .

(b) The IUCN site also provides access to online guidelines for protected area managers on the economic values of protected areas;[80]

(c) The site of the Ramsar Convention on Wetlands provides access to the Ramsar guidance the valuation of wetlands;[81]

(d) A recent report prepared by the World Bank in cooperation with The Nature Conservancy and IUCN, on assessing the economic value of ecosystem conservation, is also available online.[82]

3. Enhancing global capacities: international information systems and databases

A number of other web sites contain valuation data for more advanced practitioners. Most noteworthy is a range of sites developed to facilitate benefits transfer, such as the Environment Valuation Reference Inventory (EVRI) database sponsored by number of countries and hosted by Environment Canada, as well as some other initiatives discussed in paragraph 99 above. The inventory provides a compilation of primary valuation data from studies conducted in different countries around the world. The basic idea is for the user to define a resource to be valued (e.g. a rare species or a water body), and to search the database for studies that have generated similar information. If the studies are suitably similar, then the database provides the basis of a transfer value that fills in an information gap at the site of interest.

Benefits transfer is still under development, with numerous academic research exercises focused on the validity of transferring benefit or willingness-to-pay unit values or the statistical functions that predict these values. Nevertheless, the use of value transfer seems to be an appealing way to advance the use of valuation information in particular in resource poor countries where time and resource constraints will typically prevent extensive primary research in many decision-making situations.

Existing databases contain a variety of studies from different developing countries, but are not specifically tailored to developing country needs, either in terms of the likely valuation studies included, or in terms of the required modifications, for instance, exchange rates and currency deflators, needed to translate values for use. Therefore, a useful collaborative initiative could be to further develop existing transfer databases and to increase cooperation among database providers with a view to increase compatibility and inter-operability, such as through the establishment of common criteria for auditing valuation work, standardized coding procedures, etc.[83] Access fees should not represent a substantial hurdle in order to ensure maximum use of the databases in particular by decision-makers and researchers in developing countries.

C. FOSTERING RESEARCH

As explained above, considerable progress has been made in the last decades in developing reliable valuation tools and protocols for their application, in particular on stated-preference techniques and benefit transfer. However, challenges for further research and development also remain, in particular

80 http://biodiversityeconomics.org/valuation/topics-34-00.htm .
81 http://www.ramsar.org/lib_valuation_e.htm . See also de Groot 2006.
82 http://www-wds.worldbank.org.
83 See Lantz and Slaney 2005 for further discussion.

with regard to the conditions for validity and robustness of the benefits transfer approach. Furthermore, further research directed at the development of a biodiversity adjustment for national accounting seems to be another important means to have biodiversity losses more reflected in macroeconomic discourse.

It was also explained earlier that valuation typically addresses ecosystem services, but not biodiversity as such. Inferring the value of biodiversity requires an in-depth understanding of the links between biological diversity, biodiversity functions, and the services that are subsequently generated. Despite recent progress made in this regard, as summarized in the Millennium Ecosystem Assessment reports, this understanding is still limited and fragmented, with many unresolved questions remaining on the specific nature of interdependencies between the structure and diversity of biotic communities, the functioning of ecosystems, and the generation of ecosystem services under different states of nature or environmental conditions. Further national and international research in addressing these important questions, including research cooperation at the international level, is therefore crucial. The involvement of all relevant stakeholders, including biodiversity-dependent industry, should be ensured, as it will gear research towards developing practical mechanisms based on plausible, realistic situations. New insights on the relationship between changes in biodiversity, for example through sudden shifts in ecosystem equilibria, and the generation of ecosystem services may lead to the further improvement of existing tools as well as to the development of new tools and methodologies for the valuation of biodiversity and ecosystem functions.

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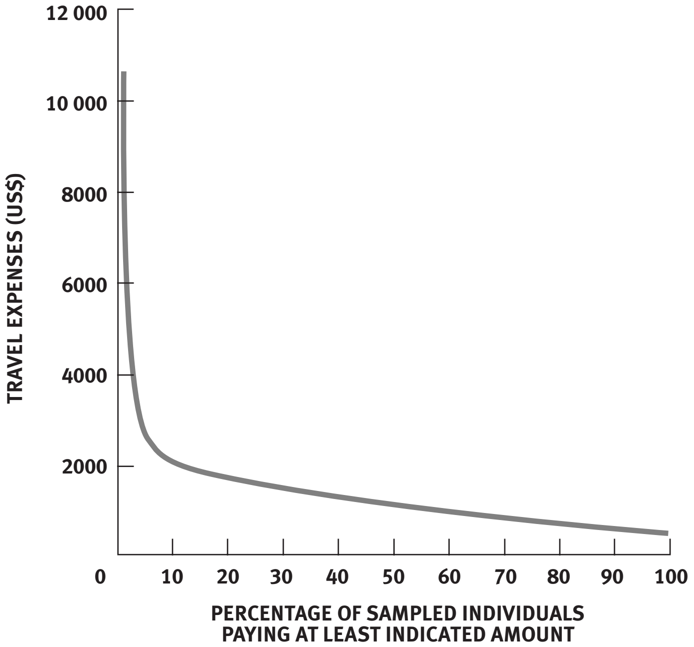
ANNEX: VALUATION STUDIES

CASE STUDY I: RAINFOREST ECOTOURISM, COSTA RICA

Region: Latin America and the Caribbean (Costa Rica)
Ecosystem valued: Tropical Rainforest
Method employed: Travel Cost

Costa Rica is home to a large variety of ecosystems, including important types of tropical forests. Over a quarter of the country is made up by parks and protected areas, but deforestation remains to be an important problem. Ecotourism has become ever more important in recent years, with the highest number of tourists visiting from the United States. Many tourists identify seeing nature as a primary reason for visiting Costa Rica. The private Monteverde Cloud Forest Biological Reserve was one of the four major ecotourism destinations at the time of the study, an example of the rare and endangered tropical cloud forest.



The study uses the travel cost method to estimate part of the value of the rainforests in Costa Rica and the Monteverde Reserve in particular. The value attributed to the forests by American eco-tourists is measured through the amount they are willing to spend on travel to the site.

The researchers surveyed a representative sample of 320 visitors to the Monteverde Reserve. They were asked for socioeconomic data, their travel expenses, and about other destinations in Costa Rica they were visiting. 240 respondents were visiting from the US, and of those 176 could provide full travel expense information.

As the pattern of airfare was found to be almost independent of distance, and as the average travel costs from different US states were almost equal, researchers eventually decided to treat all visitors as coming from the same point of origin.

The issue of multi-purpose visits was also considered. It did not cause any major disturbance of the data, since the survey of visitors to Monteverde showed they were visiting purely for ecotourism. National data on ecotourism provides numbers on visitors that came exclusively for ecotourism.

The demand curve for ecotourism was derived by evaluating the percentage of visitors that would be willing to pay at least a given amount in order to visit the park. The sample of visitors to the Monteverde reserve was used as a proxy for US tourists visiting all Costa Rican Parks and reserves. A specific value for the Monteverde reserve was estimated by adjusting the average consumer surplus per visitor for the percentage of time in Costa Rica that was spent in this particular reserve (according to survey responses). It was then multiplied by the number of US visitors to Monteverde (17,200).

The average annual valuation per person of protected areas in Costa Rica was calculated to amount to US-$1150. Multiplying this with the annual number of US ecotourists to Costa Rica (59 400), gave an overall value of US-$68 million.

The yearly US ecosystem value of Monteverde Cloud Forest Reserve amounts to $4.5 million. A previous study (Tobias and Mendelsohn 1993) had estimated domestic ecotourism value for Monteverde at around $100,000 per year in 1988. The comparison showed that US ecotourism value is much higher, which is likely due to a lack of equivalent substitute sites and higher incomes.

The study also derived a concrete policy conclusion. In order to transfer more of the benefits from parks and preserves to the less wealthy host countries, the authors recommend to use higher entrance fees. According to the estimates, entrance fees of up to $40 could be charged, which implied a significant increase of the fees that were applied at the time of the study, which ranged from $5-10.

Menkhaus, Susan; Lober, Douglas J. (1996): "International Ecotourism and the Valuation of Tropical Rainforests in Costa Rica" *Journal of Environmental Management* 47, 1–10

CASE STUDY II: MONTEGO BAY CORAL REEFS, JAMAICA

Region: Latin America and the Caribbean (Jamaica)
Ecosystem valued: Coral Reefs
Method employed: Contingent valuation, cost-based approaches

Coral reefs are one of the most diverse ecosystems on the planet. They provide a multitude of services to local communities and other users, acting as breeding ground for fish and as coastal protection. In many cases, they also provide considerable recreational value.

The Montego Bay Marine Park, Jamaica, established in 1991, has been under scrutiny both by biologists and economists since it has been transferred to private management through a trust in 1996. The Marine Park had been threatened by a multitude of adverse influences. These include pollution from ships and from shore, over fishing, and coral bleaching. At a number of popular diving sites, the high number of divers also contributes to the damage to reef vegetation.

The study sought to determine the most important elements of the total economic value of the coral reefs in Montego Bay. It looked at the values generated by fisheries, the tourism sector and the benefits of coastal protection. Contingent valuation was employed to estimate non-use values and options for biological prospecting.

A socioeconomic study was conducted as a first step in order to identify the main stakeholder groups and their concerns. This collection of pertinent data was a helpful contribution to the management of the Montego Bay Marina Park, but is also an important prerequisite for the design of the contingent valuation questionnaire. It is important that the socioeconomic situation of respondents be known in order to avoid various biases.

A conscious choice was made to exclude some of the smaller and less policy relevant types of use[84] This strategy is applied by many valuation studies, since it is often not necessary to explore the full range of services of a given ecosystem in order to have an influence on conservation policy. This is the case when the benefits associated with the most important ecosystem services are already high enough to tip the balance towards conservation. Hence, only recreation and tourism, near shore fishery, and coastal protection were addressed. A number of government, NGO, academic and consultant's documents and databases were used for collecting numbers on tourist visits, fishing activities and coastal real estate values. To arrive at net values, all costs incurred to businesses profiting from coral reefs were deducted from their gross revenue. The remaining numbers were then transformed to net present values. Different discount rates were experimented with, with a large influence on the results. A discount rate of 10% was used for calculating the cost to the tourism industry and to local fisheries (in terms of benefits lost) if the coral reefs of Montego Bay were destroyed.

Coastal protection was analyzed as the only quantifiable indirect-use value. The study used the real estate values of those properties under risk of erosion as a basis for the calculation of the value of coastal protection. The lack of certainty as regards the time it would take for a coral reef to loose this function if severely damaged, and as regards the likelihood of extreme weather occurrences in the future, is a challenge for the estimation of this value.

For the contingent valuation study, in-person interviews were conducted with 1058 respondents, including both tourists and local residents. Respondents were given information on the current situation of Montego Bay Marine Park and the human pressures it faces, as well as a general introduction to the concept of marine biodiversity. Several checks were built in the questionnaire in order to eliminate

84 The trade in aquarium fish was excluded because of its relatively small revenue, whereas the "value of coral reefs as a record of natural historical events" was considered not sufficiently policy relevant, and difficult to quantify.

bias, and the non-use and use values were separated as much as possible in order to avoid double counting. An open-ended format was chosen, with explanation of answers and reactions to changed scenarios sought.

For bioprospecting values, a statistical model was adapted from other studies, based primarily on the average market value of a new substance, the likelihood of discovery, and the density of species at Montego bay. Specific assumptions were made on the share of the rent from genetic diversity that would be captured by Jamaica.

SUMMARY OF VALUATION RESULTS—MONTEGO BAY CORAL REEF

	Benefit	Price*	
	NPV (MM$)	**MM$/%**	**MM$/ha**
Tourism/recreation	315.00	7.33	17.18
Artisanal fishery	1.31	0.03	0.07
Coastal protection	65.00	1.51	3.54
Local non-use	6.00	0.24	0.56
Visitor non-use	13.60	0.54	1.28
SUBTOTAL	**400.91**	**9.65**	**22.63**
Pharmaceutical Bioprospecting (Global)	70.09	0.23	0.53
TOTAL (GLOBAL)	**471.00**	**9.88**	**23.16**
Pharmaceutical Bioprospecting (Jamaica)	7.01	0.02	0.02
TOTAL (JAMAICA)	**407.92**	**9.67**	**22.68**

* Marginal benefits shown at typical current reef conditions.

The researchers also calculated the marginal costs of enhancing the protection of corals reefs, mainly through the reduction and better treatment of waste, by transferring the cost function from another study (Ruitenbeck 1998). They found marginal costs to range between $1 million to $29 million for increases in reef quality between 1% and 20%. The calculation of marginal costs made the determination of an "optimal" conservation strategy possible. The optimal improvement was found at 13% with associated expenditures of $27 million.

Ruitenbeek, Jack; Cartier, Cynthia (1999): "Issues in applied coral reef biodiversity valuation: Results for Montego Bay, Jamaica"—*World Bank Research Committee Project RPO# 682–22*

CASE STUDY III: CONVERSION OF MANGROVES TO SHRIMP FARMS, THAILAND

Region: Asia and the Pacific (Thailand)
Ecosystem valued: Mangrove Forests
Method employed: Change-in-Productivity, Replacement Costs, Benefits Transfer

Mangrove forests provide a variety of goods and services both to local communities as well as on a national and global level, ranging from the provision of wood to costal protection. They constitute a unique ecosystem, adapted to brackish water and different levels of flooding, with highly diverse flora and fauna. They also provide an important habitat including serving as breeding ground for fish.

Many mangrove ecosystems have been and are being converted to other uses, in the case of this study to intensive aquaculture, in form of shrimp farming. At the time of the study, they were legally state property, but were in practice considered open access to any users. In some cases, shrimp farming was encouraged by the government, since it yielded higher financial profits and export opportunities.

The study sought to evaluate the benefits provided by the mangrove forests around Tha Po Village, where approximately 400 ha of Mangroves, of originally more than 1000 ha, remained intact. Using both change-in-productivity and replacement cost approaches, the study covered the extraction of goods from the Mangroves as well as their importance as breeding grounds for offshore fisheries, as costal protection from erosion and as carbon sinks slowing global warming.

Local direct use values were elicited through two surveys of village households that focused on collecting data on the harvesting frequency and quantity of different forest products. Short interviews were also conducted during a village gathering. From the 131 households in Tha Po the first survey received 39 useful replies, but only 10 responses were useful in the second survey. The collected information was confirmed by the short interviews at the gathering. The main products were fish and other marine animals, non timber products such as honey, as well as wood for repairing fishing equipment. Only small quantities of firewood were gathered since the wood from the mangroves is not considered suitable to combustion. Market prices were applied to these goods where possible, and prices for close substitutes were used when no markets existed. Assuming that the harvesting is undertaken in the villagers' leisure time, the opportunity costs of harvesting were calculated as a share of the local wage, and subtracted to arrive at the exact value of goods.

In order to estimate the influence of coastal wetlands on off-shore fishery, the study used the Ellis-Fisher Model, a statistical model, with different functions tested for sensitivity. Historical data on catch rates, fishing techniques and mangrove size served as a basis for calculations.

Coastline Protection and Stabilization services were valued using a replacement cost approach. At coastal strips where mangrove forests are lost, breakwater dams need to be constructed in order to avoid severe erosion. The costs of these dams amount to \$ 875 per meter of coastline. The researchers had initial concerns of overestimate values, since replacement cost approaches have shown such a tendency, but the end results were clear enough even with this aspect excluded.

The value of Carbon Sequestration was calculated primarily through a benefits transfer. The total amount of carbon was calculated from the biomass per hectare of mangrove forests. A value of \$ 5.67 per ton of carbon was applied, based on a World Bank report on Mangroves in Malaysia. This can be considered as a lower bound, since other studies find this value to be considerably higher.

All the derived figures were transformed into their net present value (NPV) to facilitate a cost-benefit comparison with the value created by shrimp farms.

NET PRESENT VALUE OF MANGROVE FOREST BENEFITS*	
BENEFIT	**Value (US$) per ha**
DIRECT USE VALUE:	
Net income from timber and non-timber products	87.84
INDIRECT USE VALUE:	
Offshore fishery linkages	20.82 – 68.90
Coastline protection	3,678.96
TOTAL DIRECT AND INDIRECT USE VALUE	3,787.62 – 3,835.70
DIRECT USE VALUE ONLY:	
Net present value (10% discount rate)	822.59
Net present value (12% discount rate)	734.83
Net present value (15% discount rate)	632.27
DIRECT AND INDIRECT USE VALUES:	
Net present value (10% discount rate)	35,470.72 – 35,920.98
Net present value (12% discount rate)	31,686.34 – 32,088.57
Net present value (15% discount rate)	27,264.13 – 27,610.22

* All net present value calculations are based on a 20-year time line.

The values in terms of carbon sequestration were not included in the NPV since, as they occur on a global level, they could not be captured directly by Thailand.

The net present value of shrimp farms was calculated as being around $ 200 per hectare, becoming negative when the costs of restoring the forest, after shrimp farming was not profitable any more (yields tend to decline over time), was included.

Even when only including some of the use values and none of the non-use values[85], the NPV of mangrove forests is far higher than the NPV of shrimp farms, resulting in a negative NPV for conversion. In consequence, the calculations show that the conversion of mangrove forests to shrimp farms is, from society's viewpoint, economically not profitable.

Sathrathai, S. and Barbier E.B., (2001): "Valuing Mangrove Conservation in Southern Thailand", *Contemporary Economic Policy* 19(2):109-122.

85 E.g values for Tourism, local climate influences and existence values are missing in the analysis.

CASE STUDY IV: THE TOTAL ECONOMIC VALUE OF FORESTS IN SEKONG PROVINCE, LAO PDR

Region: Asia and the Pacific (Lao PDR)
Ecosystem valued: Forests
Methods employed: Change-in-Productivity, Replacement Costs, Benefits Transfer
Special features: participatory approaches

The territory of the Lao People's Democratic Republic contains high biodiversity—being part of the Indo-Burma geo-region, it is part of one of the twenty-five global "biodiversity hotspots". Sekong province is part of the Central Annamites, one of five priority regions of the WWF's Ecoregion Conservation Program in Indochina. Natural forests are an important source of subsistence and income to local communities, providing a large amount of non-timber forest products. They also provide a number of other benefits on the national or international level. Forests are however under increasing pressure, in particular from efforts to increase forestry output.

The study by Rosales et.al. seeks to estimate the total economic value (TEV), especially the local value, of a natural regeneration forest that is not greatly influenced by human impacts. The study uses a variety of methods, in particular participatory approaches along with changes in productivity and replacement costs. Benefits transfer is used to estimate some of the indirect use values such as watershed protection, biodiversity conservation and carbon sequestration.

The study includes an analysis of the social situation of local communities, highlighting significant differences in the use of non-timber forest products by different income groups. Poorer households are much more dependent on forest products as both monetary and non-monetary income, with 71 percent of their total income resulting from non-timber forest products.

The value of timber extracted in Sekong province was estimated using numbers for timber export revenues and tax earnings. Whether this level of timber extraction is sustainable and compatible with a natural regeneration forest that allows for the full use of other benefits was not addressed in the study.

Two different methods were used for the calculation of the value of non-timber forest products. "Focus group discussions" were conducted by asking villagers to estimate the amount of different products harvested per the year, including a ranking of importance. The found amounts were then averaged to per-household numbers, and the market price for these goods applied.

Participatory environmental valuation was used for a second valuation of non-timber forest products. This modified form of contingent valuation addresses one problem that is frequently encountered in subsistence economics, that is, the lack of relevance of money for local communities, which makes it difficult to express willingness-to-pay in monetary units. In order to address this problem, a good can be chosen that (i) is of high importance to local communities, (ii) is of relative homogeneity (so that it can act as a numeraire good), and (iii) has a monetary market value. In this case, rice, the main staple food, was chosen to act as numeraire. Respondents were asked to place counters on different cards representing non-timber forest products as well as on one card representing a sufficient amount of rice for the household. The number of counters signified the importance placed on that particular good. The amount of rice needed in a household can be taken from use statistics, and valued at a market price. Through the placing of markers, all other goods can be put in a relation to rice, and a price attached to them accordingly.

This technique resulted in considerably higher values than the focus group discussions, leading to a total value of non-timber forest products at $525 compared to $398. This could be explained by inaccurate answers, but, since harvest is mostly done for consumption and only small portions are traded on markets, more likely shows that the utility derived by communities from non-timber forest products is actually higher than its market value.

Indirect use values include the protection of watersheds and the reduction of floods, droughts and erosion or sedimentation. Indirect use values were calculated through estimated changes in productivity regarding farming, fisheries and hydropower facilities, as well as the "replacement costs" of having to build dams for flood control. Benefits transfer, using figures from various other case studies, was also utilized, generating generalized values for bioprospecting and carbon sequestration. The governments' conservation expenditures was taken as a revealed willingness-to-pay for biodiversity, representing a lower bound of the actual value of biodiversity.

DIFFERENT BENEFITS FROM FORESTS IN SEKONG PROVINCE

TYPE OF USE / BENEFIT	Annual Value / Benefit ($)	Annual Value / Benefit ($/ha)
Direct Uses:		
NTFP	4,906,942–6,472,725	398–525
TIMER REVENUES	605,000	13.35
Indirect Uses:		
WATERSHED PROTECTION		
FISHERIES & AQUATIC RESOURCES	135,919	0.47
AGRICULTURAL PRODUCTION	714,550	2.5
MICRO-HYDROPOWER FACILITIES	792–5,367	0.003–0.02
POTENTIAL HYDROPOWER SUPPLY	67,255,472–455,575,755	233–1,581
FLOOD CONTROL	26,597,000	92.3
BIODIVERSITY CONSERVATION		
CONSERVATION EXPENDITURES	1,887	0.07
BIOPROSPECTING	13,659–68,289	0.11–0.55
CARBON SEQUESTRATION	649,400,000	1,284

The results of the study are summarized in the table. While the study provides an overview of the magnitude of the services provided by Sekong forests, it also reflects the difficulties encountered when seeking to estimate a TEV with limited resources, such as the application of a mixture of instruments with potential overlaps and varying accuracy, in particular when estimating indirect use values. However, the figures derived for the direct use values alone indicate the high importance of non-timber forest products to local communities and, consequently, the need to take them fully into account in development decisions.

Rosales, R. et al. (2003): "Balancing the Returns to Catchment Management: The Economic Value of Conserving Natural Forests in Sekong, Lao PDR", *IUCN Water, Nature and Economics Technical Paper No. 5*

CASE STUDY V: CHANGBAISHAN MOUNTAIN BIOSPHERE RESERVE, CHINA

Region: Asia and the Pacific (China)
Ecosystem valued: Forest
Methods employed: Change-in-productivity, cost-based approaches

The Changbaishan Mountain Biosphere Reserve includes a forest ecosystem located in Northeast China, close to the border to North Korea. It was accepted into the World Biosphere Network for its outstanding ecological value, and is considered a strict reserve (the highest level of protection) under IUCN categories. It encompasses almost 200,000 ha, most of which are forested.

The study analyses the values of some functions provided by the forest ecosystem, that is, water conservancy, soil protection, CO2 fixation, nutrient cycling, pollutant absorption as well as disease and pest control. The study uses the change-in-productivity approach: drawing on biological research, it estimates the physical quantity of services provided and, in a second step, puts a price tag to these quantities. This price tag is calculated by mainly using the costs of providing the service by other means.

Forest cover slows water runoff and forest soil shows higher permeability, leading to increased water supply and an improved water quality. The function of water storage and increased year-round water supply is taken into account by using data on precipitation and evaporation in the area for calculating the runoff amount under a fully effective ecosystem function (623 mm/year), and by subsequently calculating the costs of alternatively building water reservoirs. Official figures on reservoir construction costs were adjusted by changes in labor and material costs plus maintenance costs. From the resulting numbers, a value for water conservancy can be derived by converting runoff figures to absolute amounts of water in the entire reserve.

The soil protection function provided by forests is quantified as the prevention of soil erosion. Research in Japan has shown that, depending on soil type, forests reduce erosion by 10 to 50 mm per year. As a mix of soils is present in the region, the study assumes 30mm per year. The costs of erosion are represented through the opportunity costs of not being able to use the land for timber production.

Carbon retention is calculated by using the photosynthesis equation and other chemical reactions inside plants. The different forest types that are present in the Reserve were taken into account with different biomass per hectare figures. Two alternative methods were considered to calculate the price (cost) of carbon retention. One relies on the European Carbon Tax systems to extract a value of $150/t carbon in Norway. The other, which was chosen for calculations, looks at the cost of afforestation to prevent global warming. Numbers from the FAO and several studies on the costs of planting trees are transferred to the situation in China and then converted from per-hectare figures to costs per ton of carbon (250 Yuan/ t C).

Nutrient Cycling is another important service provided. Trees accumulate nutrients from the ground and pass part of them on when shedding leaves or dying off. Natural nutrient cycling could be replaced by synthetic fertilizers, their price providing a market value for the service.

Forests also absorb and decompose other pollutants. The study only takes into account sulfur dioxide, valuing it through the cost of technological ways of controlling SO2 Emissions.

Pest control is provided by the forest through natural enemies and other interconnections of the ecosystem. In a natural forest potentially harmful insects are controlled by their natural enemies and a diverse ecosystem makes it harder for plant diseases to spread. This service is again valued through the costs of alternative measures, namely the use of pesticides.

Other ecosystem services are verbally discussed in the study considered but not included in the numerical calculations, mainly because of limitations to adequately quantify these services. In consequence, the numbers found in the study provide a lower bound of the total economic value of the forest ecosystem.

The study calculated the following annual values for the respective services, provided by the entire reserve (in million Yuan):

Water Conservancy:	156.14
Soil Protection:	2.20
Carbon retention:	292.53
Nutrient Maintenance:	43.39
Pollutant Decomposition and Pest Control:	15.85
Overall Value:	510.11

According to the study, the opportunity costs of the reserve, in form of not being able to produce timber, are in the order of 50 million Yuan. In consequence, the economic value of the reserve is at least ten times higher than the timber production value.

Xue, Dayuan; Tisdell, Clem (2001): "Valuing ecological functions of biodiversity in Changbaishan Mountain Biosphere Reserve in Northeast China" *Biodiversity and Conservatio*n 10, 467–481.

CASE STUDY VI: DRYLAND BIODIVERSITY: THE FYNBOS BIOME, SOUTH AFRICA

Region: Africa (South Africa)
Ecosystem valued: All ecosystems, focus on the Fynbos Biome
Methods employed: Contingent Valuation, Choice Experiment

The Fynbos Biome is an important component of the Western Cape Province, which holds the worldwide highest non-tropical concentration of higher plant species. Fynbos is the natural scrublands vegetation occurring in a small belt of the Western Cape, mainly in winter rainfall coastal and mountainous areas with a Mediterranean climate.

South African biodiversity is generally threatened by a multitude of factors, including invasive species, urban development and in particular climate change, the later predicted to cause more than 50% shrinkage in the current vegetation cover.

The study focused on directly assessing the willingness-to-pay for biodiversity conservation. It used contingent valuation techniques, applying the guidelines set out by the NOAA panel in order to minimize bias (see paragraphs 53–54). The questionnaire was repeatedly tested, and the actual survey was conducted through 820 personal interviews. Socioeconomic data was collected, including age, income and other factors, and analyzed using statistical models. Respondents were found to be largely representative on a regional and provincial level. Some adjustments had to be made for the national level.

In the section that addressed the willingness-to-pay for biodiversity conservation, respondents were first asked about their general interest in nature, and their knowledge of some major protected areas. This led to a question on how much desertification would reduce their likelihood to visit. The general annual willingness-to-pay towards nature conservation, through higher costs of electricity and petrol, was solicited through an open-ended question. Zero-bids were followed up to explore their cause. Respondents could then choose how they would like to see their money allocated to different biomes in SA. The questions then focused on the Fynbos Biome, inquiring into respondents' knowledge thereof.

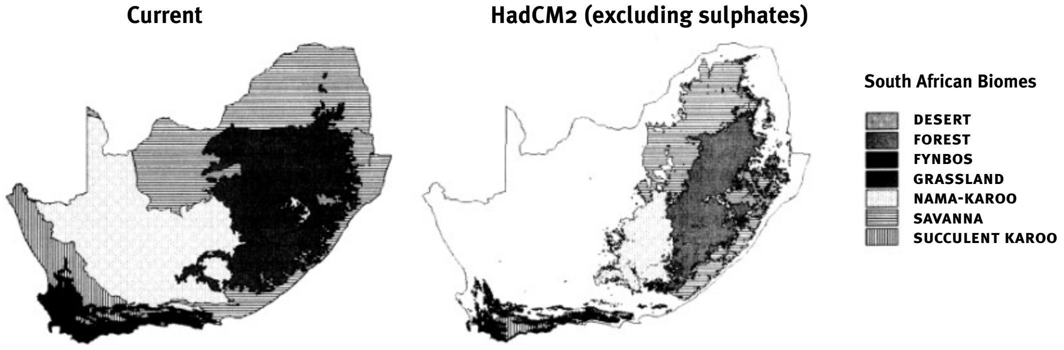
As researchers felt that the threat of climate change would have influenced replies through a sense of impending loss, it was introduced only after the first willingness-to-pay question. Respondents were shown two maps with the predicted losses in biomes (see figure) and asked whether this development were of concern to them. If there were no concerns, an explanation was sought, if there were concerns another question on willingness-to-pay followed. The researchers used a dichotomous choice format if respondents expressed a general acceptance for payment. Respondents were first asked if they were willing to pay a certain amount for the complete preservation of the different biomes. If this was refused, the willingness to pay a smaller amount for a reduction of losses was inquired (double bounded dichotomous choice). The starting amount was varied across the sample, taking five different levels. Another open ended question was used to elicit the full maximum willingness-to-pay for the 100% conservation option.

VALUES FOR THE DICHOTOMOUS CHOICE QUESTIONS IN THE DIFFERENT VERSIONS OF THE QUESTIONNAIRE, IN SOUTH AFRICAN RAND [86]

RESULTS:	VERSION 1	VERSION 2	VERSION 3	VERSION 4	VERSION 5
No loss of vegetation	1000	500	250	100	50
Intermediate loss	500	250	125	50	25
Loss as predicted (see maps)	0	0	0	0	0

Under the first, open-ended, willingness-to-pay question on nature conservation in South Africa, 76% of respondents were willing to make an annual contribution, at a mean of about $10. Projected to a provincial level, this equals a willingness-to-pay of over $8 Million per year for nature conservation. 39% of this amount would be allocated to the Fynbos Biome in the Western Cape region. On a national level an aggregate of about $58 Million would be reached. This is of comparable magnitude with government conservation budgets.

A majority of respondents were strongly disturbed by the predictions of the influence of climate change, leading 76% of respondents to favour a policy which would lead to a rise in costs for electricity, petrol, and other polluting commodities, with an overall average willingness-to-pay of $37 by the dichotomous choice questions, and $21 by the open ended question. This amounts to provincial values of $ 21–39 million, and national values between $ 161 and 262 millions.



PREDICTED LOSS OF VEGETATION IN SOUTH AFRICA. SOURCE: KIKER 2000

Besides demonstrating how contingent valuation techniques can be applied in a developing country context, this study also underlines the importance of information and education for receiving accurate WTP numbers, and how questionnaires can be built in a way to reflect the impact of information on the willingness-to-pay. In the case investigated by the study, the knowledge of the biome had a high influence, and the presentation of expected losses in case of inaction raised the willingness-to-pay considerably.

Turpie, J. K. (2003): “The existence value of biodiversity in South Africa: how interest, experience, knowledge, income and perceived level of threat influence local willingness to pay” *Ecological Economics* 46, 199–216.

86 One Rand equaled US-$0.1 at the time of the study.

CASE STUDY VII: THE FLAMINGOS OF LAKE NAKURU NATIONAL PARK, KENYA

Region: Africa (Kenya)
Ecosystem valued: Inland Waters, Wildlife viewing
Methods employed: Travel Cost Analysis, Contingent Valuation

Lake Nakuru National Park, the only Ramsar site in Kenya, is known for its spectacular number of flamingos. It also preserves a number of other rare species, such as the black and white rhino. It was established in 1972, and has since been expanded to an overall size of 188 km^2. A high number of visitors, untreated waste water and poaching constitute serious threats to the ecosystem, with the numbers of flamingoes decreasing.

The study sought to estimate the recreational value of the flamingo population, the visit to the park, and wildlife viewing in Kenya in general. It used the travel cost method as well as contingent valuation. This summary will focus on the former.

A survey of 185 visitors to the park was conducted over five months in 1991, with Kenyan residents and foreign visitors treated separately and weighed according to their share of overall visits. In-person interviews were conducted, including only adult visitors.

For the travel cost analysis, trip costs, time spent on the trip, visitors' willingness to extend their trip, and their motives for coming to the park were estimated along with the costs of visiting substitute sites. Socioeconomic data such as age, income and education were also collected and included in the demand function as independent variables. The dependent variable was the per capita visitation rate for each zone when a zonal model was used.[87] For a model based on individual observations the probability of participation was used. Different models (linear, semilog, loglinear), varying independent variables as well as different definitions for zones were used and later compared for statistical quality.

The costs for time spent on the trip were calculated as a product of travel time and the hourly wage of respondents, multiplied with a factor of 0.3. This is to reflect that the value of time while going on vacation is considered lower than the wage rate. Such a deduction is commonly used in travel cost studies.

The willingness to extend the stay was used as a proxy for site quality. In the demand function for resident visitors, a ranking of experiences in the park compared to other parks was substituted.

All different forms of the demand function were used to calculate consumer surplus and a "best" form chosen, based on consistency of the model with actual observations.

For non-residents, demand functions based on zonal observations produced results that contradicted economic theory and previous studies, therefore models based on individual observations were preferred. The model with the most consistent results calculated a benefit for all 127 non-resident visitors of about $200,000. The value per visitor per day was $75–79. Visitors stayed in Lake Nakuru National Park for 1,5 days on average, the recreational value of the park would thus be about $120. With 88,529 international visitors a year, a total non-resident value of about $ 10 million is arrived at.

For resident visitors, models based on individual observations also showed better results than ones based on zones. The same tests of statistical reliability were employed to pick the function best describing the demand for recreation at the park. It provides results of $4000-5000 for the 58 residents interviewed, about $70 per visitor. 53000 residents visited the park in 1991, receiving $3.6 to $4.5 million in benefits.

87 In the TC method, it is possible to create demand functions from individual observations, travel costs from different zones (usually geographical areas with similar travel costs) or through more complex statistical models.

The total recreational value of Lake Nakuru National Park is \$13.6–\$15.1 million by the most accurate models found. Visitors reported spending about 40% of their time on viewing and photographing flamingoes, these are thereby valued at over \$5 million.

The contingent valuation undertaken in parallel resulted in values from \$5.5million (willingness to pay) to \$12.2 million (willingness to accept) for the National Park, values from the TC model were much higher. A decision between these two values is hard to make, as both could capture different value aspects.

The study suggests an increase in entrance fees to capture a larger portion of values, especially since the models show demand to react only slightly to prices. The results were taken into account by the Kenya Wildlife Service in its long term pricing policy.

Navrud, S.; Mungatana, E.D. (1994): "Environmental valuation in developing countries: The recreational value of wildlife viewing" *Ecological Economics* 12, 125–139.

CASE STUDY VIII: MANTADIA NATIONAL PARK, MADAGASCAR

Region: Africa (Madagascar)
Ecosystem valued: Rainforest
Methods employed: Travel Cost Analysis, Contingent Valuation

The island of Madagascar is home to one of the highest numbers of endemic species in the world, with about 75% of known species not occurring anywhere else. However, with only about 17% of the original vegetation remaining, much of the ecosystem has already been profoundly altered. Due to inadequate farming and logging practices, large parts of the island have turned into deserts, and much of the original megafauna was lost to the hunting of early settlers.

Nature tourism has become an important economic factor in Madagascar, with numbers of visitors constantly rising. This provides economic opportunities for residents, especially since the government has adopted measures to share proceeds with local communities.

Mantadia National Park was established as part of the Madagascar Forest Management and Improvement Project, supported by the World Bank. The operation of the Park is based on the assumption that increased revenue from nature tourism would soon cover operating costs and provide additional profits.

The study sought to estimate the benefits to international nature tourists from the creation of this new national park, using the travel cost method as well as contingent valuation.

The travel cost analysis is based on the assumption that tourists have the option of visiting various destinations that offer similar experiences. They hence have to make a choice based on the quality and the price of the trip. A "typical trip" model was designed based on the different statistical models available, which uses data from all recreational sites under consideration to compute one single demand function. The dependent variable is the total number of nature tourism visits to all considered countries over a period of five years; the independent variables are costs, the quality of the experience as well as socio-economic variables.

The change in quality of a nature trip to Madagascar caused by the creation of a new national park would lead to a shift of the demand curve. The change in the total benefit can then be calculated by measuring the area between the old curve, the new curve and the costs.

Data were collected from two surveys. The first one was carried out in Madagascar; as it could however only acquire 87 completed questionnaires (due to a temporary reduction in tourist visits), a number of international nature travel industry experts were questioned in order to complement this information,.

The on-site survey inquired into the costs of the current trip, past or planned visits to other nature tourism destinations as well as the socioeconomic background and the willingness to pay for a new national park, to be used in the CVM study. The questionnaires had been extensively tested and reviewed, both with previous nature tourists to Madagascar in the United States and a small sample of locals.

From the question on visits to other countries, the seven most important substitute destinations were derived, namely, Kenya, India, Indonesia, Mexico, Thailand, Nepal and Brazil.

For estimating the quality of sites, tourists were first asked to rank the importance of several aspects for the overall quality of those sites.

QUALITY ATTRIBUTE	AVERAGE RATING
Exceptional beauty of the natural areas of destinations	8.22
The possibility of observing and interacting with people from different cultures	8.18
The possibility of seeing unusual animals in their natural environment	8.07
Availability of high quality guides, educational materials, and facilities for interpreting the natural areas	6.36
The total cost of the trip	5.95
Quality of transportation facilities within the country	5.39
The ability to see a number of different sites in a short period of time	5.03
Quality of the accommodations for eating and sleeping in the country	4.92
Availability of packaged tours	3.83
Amount of travel time involved in reaching the destination	3.70

In a next step, the quality of different countries regarding these aspects was determined by a survey of 22 representatives of travel agents and tour operators specializing in nature tourism to those countries.

	Destinations							
ATTRIBUTES	KENYA	INDIA	INDONESIA	MEXICO	MADAGASCAR	THAILAND	NEPAL	BRAZIL
Local guides, educational materials and facilities for interpreting natural areas	8.05	6.29	6.06	6.56	4.15	7.00	7.00	6.41
Possibility of seeing unusual animals	9.10	7.13	7.06	5.08	8.47	5.88	6.16	7.00
Accommodation and transportation services	7.97	5.67	6.83	7.67	3.80	8.65	6.47	7.74
Exceptional beauty of the natural area	8.23	7.34	8.00	6.75	6.80	6.94	8.55	7.61
Possibility of observing and interacting with people from different cultures	6.84	7.56	7.83	7.00	5.50	7.88	7.78	6.88

Travel costs for substitute sites were calculated using prices for regular airfare, and the in-country expenses as estimated by the experts.

The regression model showed all independent variables to have the expected influence, with the exception of income. This might be due to high average incomes in the sample, to a point where they might not have a significant influence on travel decisions any more.

The quality impact of the new national park was determined based on field visits. It can be expected to primarily improve nature interpretation facilities such as trails, a nature education center and trained guides. The study did not expect the new park to significantly increase the natural beauty of Madagascar or the likelihood to see rare wildlife, since this can already be achieved in existing protected areas.

Different possible improvements in interpretation facilities, ranging from 10% to 50% were entered into the model, showing a change in consumer surplus by $46 to $268 per visitor, which implied a net present value of $4–26 million.

Mercer, E.; Kramer, R.; Sharma, N. (1995): "Rain Forest Tourism—Estimating the Benefits of Tourism Development in a new National Park in Madagascar", *Journal of Forest Economics* 1:2, 239–269.

CASE STUDY IX: THE CONSERVATION OF WILD GEESE IN SCOTLAND, UNITED KINGDOM

Region: Western Europe and others (United Kingdom)
Ecosystem valued: Wild Geese
Methods employed: Contingent Valuation, Choice Experiments, Market Stall (participatory)

The study by Hanley et. al addressed a number of closely related species of wild geese with significant emotional value. With hunting of wild geese playing only a minor role today, most people derive utility either through appreciation of their beauty (direct use value), or through mere knowledge of their existence (existence value). Several species of geese in Scotland are endangered. While some protection measures are in place, calls had been made for tighter and more effective policies.

The study uses the techniques of contingent valuation (CV) and choice experiments (CE), as well as the participatory approach of a “market stall” (MS).

The market stall approach addresses the criticism that contingent valuation is taken out of a social context. It lets a group of participants act as potential buyers, while a researcher takes the part of a seller. The participants are given time to discuss among themselves before expressing their willingness to pay (WTP) and, in addition, they are given a chance to revise their WTP after two weeks. By providing a more realistic framework for decision-making, the market stall approach represents one possible way to test and improve the accuracy of CV studies.

In addition to determining overall WTP values, the study also explored the differences in WTP among areas and local vs. national populations, thus elucidating the potential distributional impacts of proposed policy changes.

Over 1200 individuals, including local residents of possible goose protection sites, visitors, and general residents of Scotland, were questioned in one-to-one interviews with the choice experiment method, 412 respondents were covered by the CV method. Questions also targeted socioeconomic information such as age and income, so as to ensure that the respondents’ sample was representative.

In order to eliminate biases or general flaws, the questionnaires were tested in pilot interviews with a limited number of participants. Extensive feedback was collected from participants after the study. While no major flaws were detected, some participants would have preferred more extensive information on the issues at stake.

For the CV a poly-chotomous choice (see paragraph 55) format was used, with variations in the initial amount asked in order to avoid potential biases. Respondents could not only express willingness to pay or not to pay, but stages in between, such as “probably would pay”. They were asked whether they would be willing to accept increases in taxes from 50 pence to 280 pounds for different protection scenarios.

The scenarios included: (i) banning all shooting of geese; (ii) preventing a 10% drop in population of all goose species (iii) insuring a 10% rise in two endangered species; and (iv) aiming at a 10% rise in all the main migratory species.

In the choice experiments participants were presented with sets of two options, each outlining certain standards of geese protection and certain costs. They could choose either or opt for the status quo. An example of such a card is included below.

SAMPLE CHOICE CARD FROM THE CHOICE EXPERIMENT	
PLEASE CONSIDER THE FOLLOWING OPTIONS	
Policy A	**Policy B**
Species protected by policy **Endangered species only**	Species protected by policy **Endangered species only**
Means of control: **Habitat management & Shooting**	Means of control: **Habitat management & Shooting**
Location: **Special reserves only**	Location: **All sites in Scotland**
Population change over 10 years: **Stay the same**	Population change over 10 years: **Moderate rise (25%)**
Price per year to you over the next ten years in extra taxes: **£10**	Price per year to you over the next ten years in extra taxes: **£60**

In order to derive implicit prices from the answers of the Choice Experiment, statistical models were used that separated the influences on WTP caused by variables such as income from those referring to preferences for certain conservation measures.

The Market Stall Valuation was conducted in two Sessions with 52 participants at different locations in Scotland. Because of the relatively low number of participants, only the options of a 10% increase in endangered species and a 10% increase in all geese species were covered. The questions asked were similar to those in the CV, with a poly-chotomous choice approach and the possible answers “would certainly (not) pay”, “would probably (not) pay”, “not sure”.

RESULTS OF THE CV, FIGURES IN UK POUNDS [89]

SCENARIO	TRIMMED MEAN WTP[88], DEFINITELY YES	TRIMMED MEAN WTP, PROBABLY YES
No shooting	6.80	14.07
Prevent 10% fall endangered	7.30	15.78
10% rise, endangered	8.70	17.36
10% rise, all migratory	15.0	25.01

The table shows the results of the CV. In the choice experiment, unsurprisingly, respondents expressed a general preference for cheaper options. Some of the highest WTP figures were found for policies avoiding the shooting of geese, up to £20 from visitors to one protected site. Visitors were

88 Trimmed means are commonly quoted in CV studies as a means to ensure that estimates are conservative. Untrimmed means are frequently skewed by a limited number of large values.
89 One UK Pound, £1, equaled about USD 1.5 in 2000.

generally willing to pay for an increase in the endangered species, to an average of £13 in some groups, whereas local residents would only be willing to pay for small increases or a prevention of losses. Locals would even need to be compensated for increases in geese population over a certain limit, having a WTP of zero. The market stall approach seemed to provide higher quality results, with fewer protest answers and a higher statistical reliability. It also came up with results that were considerably lower than those obtained under standard CV—by a factor of 3.5.

This factor was used to compute a conservative overall figure for the willingness to pay of the sample group for the conservation of wild geese. Attributing this corrected figure to the entire Scottish population results in an aggregate willingness-to-pay of £13 Million annually for a 10% increase in all geese. With estimated costs of conservation measures, incurred by farmers, of less then £100,000 per year, this yields a benefit cost ratio of over 130. Overall, the study provided evidence that comparatively small increases in the numbers of a species can be unequivocally desirable. However, larger increases may have adverse distributional consequences as utility to local residents would start to drop when going beyond a certain limit.

Scottish Executive Central Research Unit (2001): "Technical Report B—Willingness to pay for the conservation and management of wild geese in Scotland"

CASE STUDY X: THE DONAU-AUEN NATIONAL PARK, AUSTRIA

Region: Western Europe and others (Austria)
Ecosystem valued: Riverside Wetlands
Methods employed: Change in Productivity, Contingent Valuation, Replacement Costs

Plans for hydro electric development on the stretch of the Danube River between Vienna and Bratislava caused one of the most heated conflicts in the history of the Austrian environmental movement. The section of the Danube under discussion was one of the last long free flowing stretches of the river that hosts a unique and diverse wetland ecosystem.

The federal government of Austria launched a research programme with a view to provide a thorough assessment of the different options at hand. The present study, which was part of this programme, uses contingent valuation methodology in combination with several other methods. Four alternative use options are compared, two of which imply the establishment of a national park while the others favor the construction of hydro-electrical power plants. The study does in cases of doubt choose a conservative approach, hence risking an undervaluation of nature for the sake of higher credibility.

Option I proposed the establishment of a smaller national park, on public land only, without any further changes made to the river management. This option would fail to address one major threat to the wetlands, that is, the erosion of the riverbed due to upstream flow changes. In past years, the river had already dropped one meter, thereby effectively draining parts of the wetlands.

Option II proposed a larger protected area, with additional land to be bought from farmers and other property owners. It also included a hydraulic engineering project, introducing a certain kind of gravel to the riverbed ion order to reduce erosion.

Option III proposed a lower-impact version of hydroelectric use, with a power station at the border to Slovakia, which would leave a large part of the wetlands, including the most sensitive part, undisturbed.

Option IV four proposed damming the river in the centre of the most valuable wetlands, not leaving any part large enough to be considered for a national park by international standards.

The first part of the survey focused on a cost-benefit analysis including direct use values, assuming a time span of 72 years and a discount rate which at 2% was chosen rather low in order to reflect the long-term importance of environmental decisions. At first, costs and benefits of electricity production were calculated, including costs for construction, maintenance, and the emission of air pollutants during construction, valued using numbers from other studies, as well as benefits in the form of the costs of alternative means to address energy needs (a gas-fired plant, including its emissions of various air pollutants, as well as demand-side measures to enhance the efficiency of energy use).

To include enhanced energy efficiency in the analysis has an interesting effect. Since increased efficiency had then, and still has, a very high return on investment, it was shown that building a hydroelectric plant would be inefficient even without taking other cost components into consideration. This approach was not chosen, since the likelihood of it being realized was considered very low..

Benefits for shipping were taken into account for all options but the first, since it would not guarantee a certain depth for most of the year.

For the two options that proposed the creation of a national park, costs were calculated for the need for infrastructure and personnel, as well as the loss of productivity in agriculture, fishing, hunting and forestry that would result from the use restriction associated with a national park. The benefits of a national park specifically for visitors (use value), were estimated through a contingent valuation survey, cross checked with the travel cost method that found similar results.

Since the construction of hydroelectric power stations was expected to negatively influence water quality, costs of maintaining the current quality were also factored in using replacement costs.

COSTS AND BENEFITS FROM DIFFERENT USE-VALUES IN ATS [90]

	OPTION I	OPTION II	OPTION III	OPTION IV
Benefits of electricity production	None	None	11.35bn	18.24bn
Costs of electricity production	None	None	16.64bn	44.62bn
Shipping benefits	None	350-500mio/year	350-500mio/year	350-500mio/year
National Park costs	< 1.6bn	1.6 bn	< 1.6bn	< 1.6bn
Costs for hydraulic engineering	None	3.7bn	< 3.7bn	<< 3.7bn
Cost for protection of groundwater	None	None	611mio	1.44bn
Benefits of area to visitors	80 / visit	80 / visit	57 / visit	50 / visit

The net present value of all use values was then calculated, to derive absolute values for rentability. The next step, still focusing on the use values only, calculated the internal interest rate, the relative rentability of a project.

To derive non-use values, a contingent valuation study with 962 personal interviews was carried out by an independent market research institute. Project I was not included it this part, since it was quite similar to project II regarding its ecological properties. Based on about three pages of information, respondents were asked about their socioeconomic background, their general attitude towards wetlands and their conservation, their knowledge of the proposed park as well as their willingness to pay an entrance fee. They were then asked what development option they would prefer, what they would be willing to pay for it, and to give their reasons for valuing the wetlands. From these answers, put into a statistical model, the researchers deduced the influences on the WTP from factors such as income, or having been to the area before.

Regarding the net present value of the use values, Project IV took the lead with 37.7 bn ATS, followed by Project III and Project II, with Project I providing a negative value. However, project II had the highest internal interest rate with 7.76%. Since many investment decisions are made based on the return-on-investment in relative terms, this would already make the large national park option the economically best choice.

The CV study found the willingness to pay for the national park per respondent to be 329.25 ATS, with all zero-bids included. This is a very low estimate, since some part of the zero-bids can always be explained by protest bids <cross ref>. Applying this value to all Austrians and over the timeframe of 72 years would amount to a net present value of 109.5 bn ATS for Project II. Considering only 20% of this

90 One Austrian Schilling (ATS) was worth about US-$ 0,09 in 1996.

number would already lead to a net present value of establishing the national park that is higher than that of any hydroelectric option.

This study was part of a research process, which, with much support from the public, eventually led to the establishment of the Donau Auen National Park on 27 October 1996.

Kosz, M. (1996), "*Valuing riverside wetlands: the case of the "Donau-Auen" national park*", Ecological Economics 16, 109–127.

CASE STUDY XI: - SZIGETKOEZ WETLAND, HUNGARY

Region: Central and Eastern Europe (Hungary)
Ecosystem valued: Riverside Wetlands
Methods employed: Benefits Transfer

The Szigetkoez Wetland case is in many ways similar to case study X above, where valuable Danubean wetlands were eventually protected by a National Park. However, in the case of the Szigetkoez wetland, the damage to the ecosystem by a hydro-electrical power generation project had already been inflicted when the valuation study was prepared. This carried the consequence that the study had to address restoration options instead of conservation options.

The Szigetkoez wetland is again located along the Danube River, downstream from Austria where the river forms the border of Hungary and Slovakia, making management of the river a shared responsibility. Before the hydroelectrical development, the wetlands showed a very high level of biodiversity which was due to a mix of different terrains under varying influence from the river. A joint power generation project by Czechoslovakia and Hungary in the 1980s was abandoned by Hungary, but Czechoslovakia went ahead with construction of a dam. A reduction of up to 90 % in water levels resulted, leading most of Szigetkoez wetland to dry out. In 1995, the construction of an underwater weir improved water supplies in some parts of the wetland, but could not restore the regular water levels nor the patterns in changes of water levels resulting in particular from flooding. As a consequence, vegetation changed drastically, with a reduction in overall biodiversity.

Due to lack of funding and time to conduct primary research, the results from the Donau Auen National Park study (see case X above) were used for a benefits transfer study. Instead of transferring a benefit function that would include a variety of variables, the researchers decided to transfer unit values with an adjustment by income only, the assumption being that the environmental sensitivity and other factors were the same in Austria and Hungary. Hence, differences were accounted for by the difference in GDP only. The authors reviewed a number of studies from other regions; their review indicated that Hungarians did have a similar willingness-to-pay for ecosystem conservation.

The two ecosystems provided some important prerequisites for a benefits transfer. Both are located on the same river, are unique within each country, and shelter a high level of biodiversity. Both were faced with the same choice of development against protection. The fact that the choice was already made in the case of Szigetkoez is not relevant here, since it is the status of the wetland before development that is being valued.

The findings of the Austrian study of willingness-to-pay per person and year (329.25ATS) were compared to the Austrian 1993 GDP per year (265,812), showing a willingness-to-pay of 0.12% of the GDP per person. This number was transferred to the 1999 GDP of Hungary, assuming that the willingness-to-pay had not changed in the six years in between. The Hungarian GDP was corrected, taking into account an illegal economy of about 15% of GDP. This resulted in the following numbers:

PROCESS OF WTP ESTIMATION IN HUNGARY AND ITS RESULTS (1999)	
OFFICIAL GDP/PERSON (HUF) [91]	1,146,000
CORRECTED GDP/PERSON (HUF)	1,318,000
WTP IN THE % OF GDP/PERSON	0.12%
WTP/PERSON/YEAR (HUF)	1,581

91 One 1999 Hungarian Forint equals about 0.004 1999 USD.

The figure for the willingness-to-pay per person and year was then multiplied by the population of Hungary over the age of 14, the same age limit as the original study.

The fact that the Hungarian wetland is considerably larger was factored in as well, assuming that the willingness-to-pay rises proportionally with the size of the area considered. This leads to an overall willingness-to-pay for (hypothetical) conservation in the original form of HUF 16.83 bn. Another critical assumption was made for the remainder of the study, namely, that willingness-to-pay values for deteriorated conditions of the wetland could be estimated by lowering the initial figure proportional to the loss in size of the undisturbed part of the wetland.

The authors considered three options as possible future states of the wetland. One is the status quo, called variant "C" in the table below. Restoration to the original state is not considered; instead, two "meandering" alternatives are considered. Under these options, further barriers would be introduced into the riverbed, which would direct water into side channels. With a resulting degradation of 5–30 % of the wetland, depending on the project, the willingness-to-pay numbers are decreased accordingly. These figures are then compounded into a net present value, and the resulting losses in net present value compared to the original willingness-to-pay. Depending on the discount rate, the following results are found:

SUMMARY OF VALUE LOSS CALCULATED BY BENEFIT TRANSFER

	VARIANT "C"	MEANDERING VERSION	
		At small and medium water level	*At high water level*
AT 2% DISCOUNT RATE (HUF BN)	168–252	126–210	42–126
AT 3,5% DISCOUNT RATE (HUF BN)	96–144	72–120	24–72

In a next step, these numbers could be used for a cost-benefit analysis, including costs for the construction of the structures needed to redirect water flow, or the opportunity costs of letting more water through the damn.

Szerényi, Marjainé Z.; Kovács, Eszter; Kerekes, Sándor; Kék, Mónika (2001):"Loss of Value of the Szigetköz Wetland due to the Gabcikovo-Nagymaros Barrage System development: Application of the benefit transfer in Hungary" OECD Environmental Directorate, OECD, Paris.

CASE STUDY XII: TOURISM VALUE OF THE GREAT BARRIER REEF, AUSTRALIA

Region: Western Europe and Others
Ecosystem valued: Coral Reef
Method employed: Travel Cost Analysis

Australia's Great Barrier Reef is the largest coral reef system in the world, consisting of roughly 3000 individual reefs combined with hundreds of islands, covering an area of almost 350 thousand km². The reef provides a variety of services to humans, including fisheries support and coastal protection, as well as recreation. The Great Barrier Reef is known as one of the best diving destinations worldwide, attracting over two million visitors every year. It is however endangered by threats such as mining, over fishing, water pollution, and global climate change.

Because of the high number of visitors, the travel cost method becomes an obvious choice for the valuation of the reef by the present study.

Two data sources were used. One was a survey of 607 persons visiting the Great Barrier Reef Sept. 1st and Dec. 1st 2000. While this data had the advantage of being representative, it also featured the problem that many visitors purchase travel packages which include other services and experiences than visiting the reef, which makes a distinction in terms of value difficult. Some visitors also benefited from special low air fares that were not available to everyone. For these reasons, an alternative set of data was generated from one single travel agent for visitors from all regions of the world. While this enabled researches to define one "standard" trip, generating comparable data from all countries, the trips taken from different countries might vary greatly in reality. Both approaches and their ability to predict actual visitor behaviour were later tested in different models. As a third approach, a 2-stage least squares estimate was also used that drew on both sets of data and travel distances to estimate the predicted travel costs.

For deriving the demand functions, visitors were grouped by origins into 39 regions. Those were mostly entire countries, except Australia, Canada and the United States which were separated into various regions. The visitation rate per population of these regions served as the dependent variable for the demand function. This is usually done where the frequency of visitation can not be used because most visitors visit only once per year, which is the case for international Great Barrier Reef visitors.

Two different statistical models were tested by analysing how well they predicted actual visitation rates. The polynomial model was considered to better fit the data, with actual travel costs performing better than those given by the travel agent. The 2-stage least square estimate provided very good results in one model, but performed badly in the other.

The table below shows the results from the two models with the different data sets used, the annual consumer surplus estimates in million USD per year. The authors decided against using the result of the polynomial model with travel agent costs, since it included very high values for Japan and the USA that were hard to explain. This is also part of a conservative approach.

The other results range from 710 million to 1.6 billion, rewults which are consistent with the figures applied by the Great Barrier Reef Marine Authority's figures which assumes an annual value of 800 million. Discounting these annual benefits at 4%, a rate rather commonly used in envrionmental valuation, yields a net present value of between USD 18 and 40 billion. This makes the Great Barrier Reef one of the most valuable ecosystems in the world, in particular when taking into account that the valuation study only captured one aspect of the total economic value of the Great Barrier Reef, that is, not including other ecosystem services provided such as fishery nursing ground or coastal protection.. Benefits per visitor amount to USD 350-800, a value comparable to those found in previous studies regarding coral reefs, but higher than most valuations of other ecosystems.

		ACTUAL COST	TRAVEL AGENT COST	TWO-STAGE LEAST SQUARES
Log-Linear	Australia	319.7	198.7	348.2
	Japan	339.1	509.7	342.0
	North America	115.0	234.9	230.5
	Europe	545.4	502.9	548.3
	TOTAL	**1378.7**	**1585.8**	**1563.0**
Polynomial-tobit	Australia	475.0	328.8	421.2
	Japan	632.2	1306.3	205.9
	North America	41.4	560.0	58.4
	Europe	15.9	4.9	0.0
	TOTAL	**1187.3**	**2223.6**	**709.0**

The distribution of benefits is also an important result. While varying among the different models and data sets, the study shows that a share of 40% to 80% of benefits go to visitors from abroad. The authors take this as an argument for international funds to protect coral reefs, especially for the benefit of poorer countries.

Carr, Liam; Mendelsohn, Robert (2003): "Valuing Coral Reefs: A Travel Cost Analysis of the Great Barrier Reef", *Ambio* 32, No. 5.

CASE STUDY XIII: THE EXXON VALDEZ OIL SPILL, UNITED STATES OF AMERICA

Region: Western Europe and Others
Ecosystem valued: Marine and Coastal Ecosystems
Method employed: Contingent Valuation

A.VERY HEAVILY OILED SHORE BEFORE CLEANUP

B. COLUMBIA GLACIER ON PRINCE WILLIAM SOUND

When the Exxon Valdez Oil tanker ran into a submerged reef in Prince William Sound, Alaska, in March 1998, this was the largest oil spill in US history, and perceived as one of the worst environmental disasters. 11 million gallons of crude oil spread over almost 2000 km of Alaskan Coastline, killing hundreds of thousands of seabirds and other wildlife. The court case brought against Exxon for natural resource damage by the Alaskan and federal government was the first high-profile case were compensation for passive-use values, or existence values, was sought. These values were determined by a contingent valuation study, triggering a major debate on the reliability of this method that also resulted in the report by the NOAA panel (Arrow et al 1993) which first set widely accepted standards for the application of this method.

The original report to the attorney general of Alaska on the passive use value lost through the oil spill was published in November 1992. Its findings were presented again by the original authors in 2003; this summary will draw on this more recent paper.

The study already followed many of the guidelines that were later established by the NOAA panel, including for instance a discrete choice referendum elicitation format and a very accurate description of the situation and policy options.

The survey was developed over an 18 month period, including field testing, work with focus groups and a series of four pilot surveys. An extensive description of damages was included to ensure that only those damages that were actually caused by the spill would be valued by participants. Such extensive briefing however also made it harder to create a survey instrument that would be understood by respondents from all areas of society and varying levels of education. Various diagnostic checks were carried out towards this end. The wording of the survey was also reviewed to ensure neutrality, and a question asking respondents on who they thought was sponsoring the study included to check if the texts appeared biased. The study tried to be as conservative as possible, in case of doubt choosing methods and designs that would result in lower willingness to pay statements.

The interviews took about 40 minutes, half of which was used for presenting the damage and a possible program that might prevent such disasters in the future. About 1600 representatively selected households from all of the United States were surveyed. A double-bounded dichotomous choice format was utilized, offering a lower amount to those replying “no” to the first question, and a higher sum to those replying “yes”. Respondents could also change their answers at a later point in the questionnaire if they wished. At the beginning of the questionnaire respondents were asked some general questions about their attitudes towards different public goods as well as their knowledge of the spill before the objective of the study was revealed to them.

The description of the area included some pictures and maps of Prince William Sound before and after the spill, showing the extent of the spill as well as clean-up measures. Scientific estimates on the duration of the injuries were presented, considering it likely that most of the effects would wear off over a few years.

BIRD SPECIES AFFECTED BY THE 1989 ALASKA OIL SPILL

	IN THE ENTIRE SPILL AREA	
SPECIES	**NUMBER OF DEAD BIRDS RECOVERED (ROUNDED)**	**ESTIMATED POPULATION BEFORE THE SPILL**
Murres	16 600	350 000
Sea ducks	1 150	100 000
Murrelets	1 150	50 000
Cormorants	1 050	30 000
Pigeon Guillemots	500	20 000
Kittiwakes	400	100 000
Grebes	350	8 000
Loons	300	3 000
Storm-petrels	300	300 000
Fulmars	250	150 000
Gulls	200	100 000
Bald Eagles	100	5 000
Other sea birds	250	300 000
TOTALS	**22 600**	**1 516 000**

Estimated numbers of killed animals were also given. Pictures of specific animals affected by the oil spill were shown. However no pictures of killed or harmed birds were shown to keep estimates conservative and reduce impulsive high bids. Respondents were also informed that large bird kills could occur naturally and that none of the affected species were endangered by extinction.

The description then turned towards a description of a program that was expected to reduce the likelihood of such injuries. The plan introduced the concept of escort ships to accompany tankers through Prince William Sound. These would be able to prevent spills by pushing or pulling tankers away from obstacles as well as contain possible spills with the suitable equipment.

The valuation was then carried out, with three different sub samples being presented with different payment options in the dichotomous choice format.

PROGRAM COST BY VERSION AND QUESTION			
VERSION	A-15	A-16	A-16
A	$10	$30	$5
B	$30	$60	$10
C	$60	$120	$30
D	$120	$250	$60

Other questions seeked to elicit why respondents had chosen amounts the way they did, and if they had accepted the scenario as it was presented. They were finally given the chance to reconsider and change their choice on the original valuation question.

A statistical model with a valuation function was then constructed using the replies, and sensitivity tests carried out thereon. USD 2.8 Billion (in 1990) were found to be the lower bound of willingness to pay estimates, and the lost passive use value. However, even very conservative calculations found some significantly higher results.

The State of Alaska and the US Government settled their lawsuits agains Exxon for USD 1 billion, in addition, Exxon spend approximately 2 billion on clean-up activities A class action lawsuit was decided in December 2006, awarding USD 2.5 billon of punitive damages, half of the original fine. These figures are consistent with the estimate of the valuation study. A program similar to that described in the study, using escort ships, has been introduced and in one case prevented a possible spill.

Carson, Richard; Mitchell, Robert; Hanemann, Michael; Kopp, Raymond; Pressers, Stanley; Ruud, Paul (2003). “Contingent Valuation and Lost Passive Use: Damages from the Exxon Valdez Oil Spill”, *Environmental and Resource Economics* 25(3): 257–286.