

OUTPUT II

BENEFITS AND COSTS OF FOREST BIODIVERSITY: ECONOMIC THEORY AND CASE STUDY EVIDENCE

FINAL REPORT

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SUMMARY

This report examines the available case study evidence on the benefits and costs of conserving forest biodiversity. The objectives are: to review the benefits provided by forest biodiversity and the costs of conserving it; to assess the extent to which these values can be used to aid decision making; and to identify key information gaps as guide to future research.

The first chapter of the report presents a framework for analysis. We look at how forest biodiversity contributes to human welfare through the direct provision of resources, the maintenance of ecosystem functions, and the protection of the resilience of the ecosystem as a whole. This is then linked to different types of economic value, using the concept of Total Economic Value. We examine the components of economic value provided by forest biodiversity, including direct use values such as the harvest of non-timber forest products and recreational uses of forest areas; indirect use values such as watershed protection and carbon sequestration; and non-use values such as awareness of the existence of biodiversity in general, or of particular species.

In addition to the different types of economic benefit, we consider the costs incurred in conserving forest biodiversity. These include the opportunity costs of forgoing alternative uses of the land and the implementation costs of conservation programmes. We also consider the issues involved in comparing the costs and benefits of biodiversity protection, such as distributional impacts, if different groups gain or lose from a course of action, or how comparisons can be made when costs and benefits occur in different time periods. Finally, we discuss the methods used in the case studies to quantify the benefits and costs of protecting forest biodiversity when market values are not available.

The second chapter presents the findings from a review of over 200 case studies on the benefits and costs of forest biodiversity conservation. The evidence on benefits is organised according to different types of use and non-use value. We discuss the conclusions that can be drawn from the case study material regarding the welfare impacts arising from the enhancement or loss of forest resources or forest diversity in different ecosystem types and different geographical locations. This is followed by an assessment of the case study material with respect to the costs of conserving biodiversity in different forest types and different locations. We also look at the extent to which the costs and benefits of forest biodiversity conservation can be compared, in order to estimate the net benefits of conservation. We conclude with a discussion of which questions can be readily answered with the information that is currently available, and where further research is required to make better decisions about forest biodiversity conservation.

Major conclusions include the following:

- Non-timber forest products and bioprospecting values are rarely sufficient to justify forest conservation on their own but are readily captured and thus will be reflected in forest land use decisions, provided that property rights are insecure;
- Recreation and watershed values will be captured locally, where they are significant, through tourism markets and payment for ecosystem services;

- Carbon values can increasingly be captured through the market, although biodiversity co-payments may be needed to avoid perverse incentives that favour carbon-rich, biodiversity-poor forests;
- Non-use values of forest biodiversity are large and largely uncompensated, due to free-riding; government intervention is required to capture non-use values and turn them into cash flow for conservation;
- Conservation costs are generally low where non-use values are high, creating an opportunity for significant welfare gains from increased conservation investment in some areas;
- Priorities for further research include the impact of marginal changes in forest diversity and economic values, the determinants of non-use value (especially in developing countries), and more use of spatial cost-benefit analysis to identify optimal conservation strategies.

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CHAPTER 1

A FRAMEWORK FOR ANALYSING THE COSTS AND BENEFITS OF CONSERVING FOREST BIODIVERSITY

CHAPTER 1

A FRAMEWORK FOR ANALYSING THE COSTS AND BENEFITS OF CONSERVING FOREST BIODIVERSITY

1.1 Introduction

According to the FAO Forest Resource Assessment (2005), deforestation is taking place at a rate of about 13 million hectares per year, mainly due to conversion to agricultural land. This includes a net loss of primary forests of around 6 million hectares per year (roughly a fourth of the UK's total land area). The greatest losses in forest cover are occurring in tropical areas of South America and Africa, which has important implications for biodiversity loss since half of all species are thought to be found in tropical forests (Wilson, 1992).

Deforestation and the associated losses in biodiversity occur for a number of reasons including perverse policy incentives, insecure property rights, or high discount rates among poor households (Shively and Pagiola, 2004). However, one of the underlying common denominators across all these causes behind forest biodiversity decline has to do with the non-market (or public good) nature of some of the benefits associated with conserving such resources.¹ This results in either imperfect or even totally absent markets for the provision of these benefits. Such cases of market or policy failure lead to the assignment of diminished if not negligible monetary value to the benefits derived from forest ecosystems, which in turn results in their under-representation in decision making processes over alternative uses of forest land undertaken by both public and private (or individual) agents. Hence, gaining an improved understanding of the economic costs and benefits of conserving forest biodiversity is a first step towards addressing the root causes of market failure and internalising the social returns of land use decisions.

Costs-benefit analysis (CBA) is a structured set of methods for comparing the benefits and costs associated with the provision of different levels of market and non-market goods and services, such as those derived from forest biodiversity. CBA is a tool-kit that operationalises the logic inherent in the main normative decision making criterion in modern economic science, namely the potential compensation Pareto criterion, which compares alternative policy options on the basis of whether they lead to an efficient allocation of resources. Put simply, a Pareto efficient policy change is that in which the net winners can potentially compensate the net losers from the change, and no other policy (or resource allocation) results in a greater overall level of utility. As the CBA apparatus is grounded on a utilitarian ethical premise, this requires the monetisation, aggregation (across space and time) and comparison of benefits and costs of policy actions. This process entails several issues that have to do with understanding the very nature of the benefits and costs included in these calculations, issues related to the measurement techniques employed to monetise benefits and costs, as well issues that concern the comparison (across time and space) of the resulting values.

¹ This refers to the non-excludable and non-rival nature of many biodiversity benefits.

This chapter discusses the aforementioned issues in the context of comparing the costs and benefits associated with conserving forest biodiversity. In doing so we aim to provide a conceptual framework for the compilation, classification and interpretation of a series of empirical case studies on the costs and benefits of forest biodiversity that is presented in the next Chapter. We first look at how forest biodiversity contributes to human welfare through the direct provision of resources, the maintenance of ecosystem functions, and the protection of the resilience of ecosystems as a whole. This is then linked to different types of economic value using the concept of Total Economic Value. We examine the elements of economic value provided by forest biodiversity, including direct use values such as the harvest of non-timber forest products and recreational uses of forest areas; indirect use values such as watershed protection and carbon sequestration; and non-use values such as awareness of the existence of biodiversity in general, or of particular species. In addition to the different types of economic benefit, we consider the costs incurred in conserving forest biodiversity. These include the opportunity costs of forgoing alternative potential uses of the land and the implementation and monitoring costs of any particular conservation programme. We also consider the issues involved in comparing the costs and benefits of biodiversity protection, such as the distributional impacts if different groups gain or lose from a course of action, or how comparisons can be made when costs and benefits occur in different time periods. Finally, we discuss the methods used in the case studies to quantify the benefits and costs of protecting forest biodiversity when market prices are not available or reliable.

1.2 Defining and measuring biodiversity

In order to discuss the costs and benefits of conserving forest biodiversity, it is necessary first to consider what biodiversity is and why it is important. The Convention on Biological Diversity defines biodiversity as:

‘the variability among living organisms from all sources, including, *inter alia*, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems’ (CBD, 1992).

This definition highlights the complexity that arises from the many dimensions of biodiversity. These various dimensions make the measurement of levels or changes in biodiversity an especially challenging task.

First, as stated above, biodiversity can be measured within species, between species, or across ecosystems. Species diversity is the most commonly used measure of biodiversity, and the concepts are often assumed to be synonymous (Hooper et al, 2005). Given that it is rarely possible to catalogue the distribution and abundance of all species within an area, three scale-based measures of species diversity are commonly used: α -diversity is a measure of the number of species in a given area; β -diversity indicates the difference in α -diversity across ecosystems in a particular area; and γ -diversity measures the overall diversity within a large region or landscape.

In addition to these measures of diversity, other indices have been developed to reflect the evenness of species distribution across an ecosystem (e.g. the Shannon index) or measures of genetic distance between species (e.g. Solow et al, 1993).

In addition to species diversity, biodiversity consists of variability at the genetic and ecosystem levels. Genetic diversity within species can be measured on the basis of the difference between individual genes (allelic diversity), differences in the characteristics of individuals within a species (phenetic diversity), or by examining DNA sequence variation. Genetic diversity may be reduced as a result of a general decline in the population of a species. This is because it cannot be recovered even if populations are subsequently restored. Alternatively, selective breeding of crops or livestock by humans may reduce genetic variability and increase susceptibility to pests or disease (Pearce and Moran, 1994).

Measurement of ecosystem diversity is complicated by the inter-relationships between ecosystems in different locations and at different scales. This means that it is not always possible to delineate clear boundaries between ecosystems in order to measure their diversity, although attempts have been made to define areas that have particular internal linkages and common ecological characteristics e.g. WWF Priority Ecoregions or Conservation International's Biodiversity Hotspots.

For the purposes of this report, simple measures of species richness are mainly used on the grounds that they are widely available. However, where other dimensions of biodiversity are relevant to particular ecosystem functions or economic values, then these will also be considered. In addition, we also discuss the implications of species or ecosystems that are particularly rare, threatened, or important in some other way, for example “keystone species”, which may be important for the maintenance of an entire ecosystem, as well as charismatic or totem species held in special esteem by some people.

1.3 The importance of biodiversity

The links between forest biodiversity and human welfare may be analysed using the framework developed for the Millennium Ecosystem Assessment (MA 2005), which distinguishes four categories of “ecosystem service”, namely “provisioning, cultural, regulating and supporting” services. The first of these refers to the obvious value of biodiversity in directly providing resources for human use, such as food, timber or medicinal plants. The second category of cultural services includes less tangible benefits such as spiritual and cultural values, as well as enjoyment through recreation and tourism. The two remaining categories of ecosystem service affect human welfare more indirectly. Regulating services refer to benefits from the regulation of ecosystem processes which enhance economic productivity, protect economic assets and secure human health. Examples include the regulation of climate, floods, disease and water quality. Finally, biodiversity provides so-called supporting services that enhance the resilience of ecosystems, and as such constitutes a form of insurance against the loss or collapse of the ecosystem and the benefits it delivers. Each of these is discussed in turn below.

1.3.1 Provisioning services

The components of biodiversity include a range of goods and services that are valuable to people. Some of the commodities derived from forests are not obviously dependent on diversity per se, for example yields of timber are generally higher from forests with single species and single age classes. In contrast, other harvested resources may be more easily attainable or more abundant in relatively diverse forests. For example, more diverse forests may allow communities to collect a range of useful plants for different purposes. Diverse forests can also provide genetic information that is used for research into new agricultural crop varieties or the development of medicinal products.

1.3.2 Cultural services

Biodiversity can also provide less tangible cultural services, for example by contributing to cognitive development, cultural traditions or spiritual inspiration. Recreational enjoyment is another service that may be provided directly by ecosystems with high biodiversity. This is particularly clear in the case of tourism for wildlife viewing, where tourists appear to gain more enjoyment from viewing more species. More generally, people may consider natural landscapes to be more attractive or more interesting where diversity is greater.

1.3.3 Regulating services

A third way in which biodiversity contributes to human wellbeing is through its role in maintaining ecosystem functions. The relationship between biodiversity and ecosystem functions is not fully understood, but there is some evidence to suggest the form it might take. Experimental and observational studies tend to show ecosystem function initially increasing with species richness, eventually reaching a plateau or even declining at higher levels of biodiversity (Thompson and Starzomski, 2006).

However, species composition may be more important than species richness for certain ecosystem functions. This is because the specific traits of the dominant species in an ecosystem tend to determine the ecosystem's processing of matter and energy (MEA, 2005). Hence it is often necessary to conserve the biological composition of ecosystems as well as the total number of species.

In addition to species composition, species interactions are important for ecosystem functions. Specifically, the complexity of inter-species relationships within ecosystems means that if linkages between species are interrupted by changes in the presence or abundance of individual organisms, then certain ecological processes may be affected. Alternatively, single "keystone species" may in some cases be critical for the continuation of the ecosystem as a whole.

1.3.4 Supporting services

A final way in which biodiversity contributes to human wellbeing, and indeed survival, is through its influence on the resilience of ecosystems. Diversity within ecosystems increases the likelihood that they can recover from external shocks and stresses (Holling et al, 1994, cited in OECD (2002)). As discussed by Pearce (2002), this relationship between diversity and resilience also applies at other levels. For example, genetic diversity enables a species to adapt to changes in external conditions. What is less well understood is the extent to which certain species may be more important than others for driving change and adaptation. Given the high degrees of uncertainty about whether some species could be redundant, maintaining diversity in general may be a more reliable way of ensuring ecosystem resilience.

1.4 Economic value of forest biodiversity

The previous section discussed the broad categories of benefits derived from diverse forest ecosystems. We now turn to discuss how these benefits acquire meaning in economic terms. In this section we discuss how and to what degree the benefits of forest biodiversity can be expressed in terms of economic values. In doing so we will clarify what is meant by economic as opposed to other forms of value, as well as identifying which of these benefits can be meaningfully incorporated in a structured comparison of costs and benefits.

1.4.1 Total Economic Value

In economic terms, a specific benefit or flow of services derived from an environmental resource - such as a forest - has value if individuals are willing to make trade-offs between this service and all other available goods and services. In other words, economic value is an anthropocentric and relative concept that can only be applied to goods or services over which individuals are willing to make trade-offs. Note that monetisation is not an integral part of the concept of economic value. However, expressing benefits in terms of monetary values is a useful convention as it allows us to express trade-offs using a single metric, namely money, which in turn facilitates the aggregation and comparison of costs and benefits in a CBA framework.

If there are forest-related services for which trade-offs are not possible, then there is little meaning in saying that they have economic value. This is not to say that such services have no worth, but simply that they cannot be expressed in economic terms and hence cannot be included in the calculus of cost-benefit analysis. Instead, to the extent that such non-economic values are identified as important to society, they can (and should) be included in public decision making via other means (see Section 1.4.4).

Despite some limitations regarding which ecological services can be considered as having economic value, it nevertheless remains a powerful concept. For example, forest biodiversity may be associated with various types of economic value through its direct provision of goods and services, its impacts on the functioning of forest ecosystems, and its role in maintaining the resilience of those ecosystems. In addition, for many

individuals, forest biodiversity has value simply by virtue of its existence, quite apart from any particular services it provides.

A commonly used conceptual framework for decomposing the separate elements of economic value for either biodiversity or specific biological resources is the notion of Total Economic Value (TEV). This identifies the various ways in which biodiversity provides flows of goods and services to humans, which in turn have an impact on their welfare. TEV consists of use values, which may be direct or indirect, and non-use values.

In the case of forest biodiversity, TEV incorporates the following values (IIED 2003) (see Figure 2):

- Direct use values: these would include timber and non-timber products harvested from forests; genetic information from forest biodiversity, which may be used as an input to agricultural or pharmaceutical research; or the enjoyment obtained from recreational activities in diverse forest landscapes.
- Indirect use values: through its impact on the functioning of forest ecosystems, biodiversity provides indirect services such as watershed protection or carbon sequestration, which ultimately have an economic value.
- Non-use values: as well as values obtained through the use of forest biodiversity, individuals may place value on forests that they will never use, either because they value the knowledge that others elsewhere or in the future can use the resource, or because they gain satisfaction from the awareness of the continuing existence of forest biodiversity in general, or of specific species.

There is also a further element of TEV that relates to both use and non-use values:

- Option value: this is the utility that an individual obtains from knowing that the resource in question will be available either for their own use or for the use of others in the future.

TEV measures ‘total’ value is the sense that it is the sum of individual components of value. However, there are a number of ways in which it does not measure the entire value of biodiversity or biological resources. These are discussed below.

1.4.2 Total value versus marginal changes in the resource

TEV is typically used as a framework to measure marginal (small-scale) changes in the stocks of biodiversity and the resulting flows of goods and services. While it may be feasible to estimate the welfare impacts of a partial reduction in the area of particular forests, or a decline in their quality, it is much less clear how to assess the welfare impacts of the loss of all forests in a country, or at a global scale. Another reason why it is more appropriate to estimate the value of marginal changes in biodiversity, in particular, is that there may be critical thresholds for the level of diversity, below which an ecosystem no longer functions. In such cases, a large reduction in biodiversity may have discontinuous and unpredictable impacts. Fig. 1 (from Turner et al, 2003) shows how continuous, diminishing marginal values may exist for small changes in the flow of

services from a natural resource (e.g. from Supply A to Supply B in the figure), until a critical threshold is reached, at which point it is no longer possible to obtain meaningful economic values.

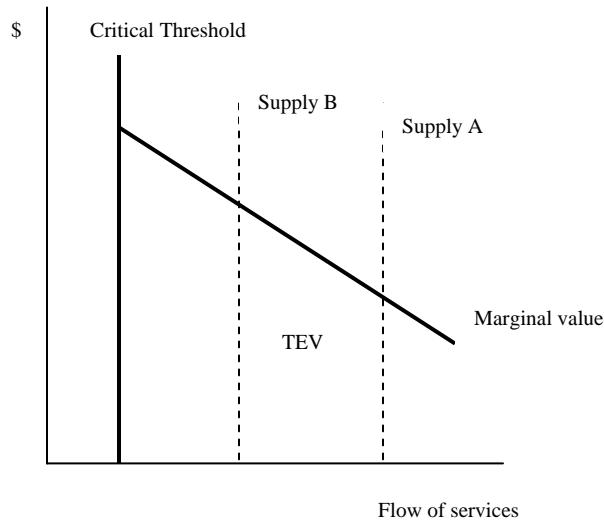


Figure 1 Valuing marginal changes

Marginal changes are also used for economic valuation because they are most relevant for decision making purposes. We mentioned above that economic values are essential for undertaking cost-benefit analysis, in order to choose between alternative projects or policy options. Additional ways that economic values may be used for policy purposes include the quantification of changes over time in stocks and flows of natural resources for the creation of environmental accounts; assessing damages resulting from industrial accidents or other events; determining appropriate levels of environmental taxes or subsidies; and setting overall policy priorities. In all of these cases, marginal values are required, rather than the total value of a particular resource.

In practice, most of the case studies collected for this report either measure the value of a change in the quality or quantity of particular forest resources in a country or region, or a small change in the area of all forests in a country or region. Some estimate the values arising from an entire forest but, to the extent that the area considered is generally small in relation to the total forest area of the country or region, it may be appropriate to consider the existence or absence of the whole forest to be a marginal comparison with respect to total forest area.

1.4.3 Primary life support functions of biodiversity

One element of the value of biodiversity that is not normally included within the TEV framework is the contribution of biodiversity to the continued functioning of a healthy ecosystem. These 'primary life support functions' may not be captured by the sum of the values of individual goods and services, and may be particularly difficult to observe or

measure (Pearce and Moran, 1994). The impact of declining biodiversity on ecosystem functionality is likely to manifest itself as a gradual loss of resilience, the outcome of which may only be observed if external changes lead to ecosystem collapse. As in the case of non-marginal external changes, this means that the costs of biodiversity loss will be discontinuous and unpredictable. In this situation, part of the value of conserving biodiversity is the insurance provided against the possibility of ecosystem collapse.

To some extent, the role of biodiversity in maintaining ecosystem resilience in the face of external shocks can be considered a form of option value or quasi-option value. The former refers to the value of retaining the option to enjoy a known use of a resource in the future, and is included within TEV. The latter is the value of retaining the option to take advantage of potential new information about or new uses of a resource, which may become available in the future. Assessing quasi-option value involves a comparison between one scenario in which irreversible change occurs, and another scenario in which forest biodiversity is retained in a reversible state while knowledge about costs and benefits increases over time (van Kooten and Bulte, 2000). Option value can be estimated using stated preference techniques (see Section 1.7.1 below). Quasi-option value, on the other hand, is difficult to estimate because there is no obvious way to assess what additional information may be obtained by waiting or how useful it might be. In other words, the context is one of uncertainty rather than risk, hence it is not possible to use expected values because probabilities are unknown.

1.4.4 Intrinsic values of biodiversity

As stated above, the TEV framework only includes economic values and thus values of biodiversity that can be quantified in monetary terms. However, there may also be intrinsic values that some individuals place on biodiversity for cultural, historic or symbolic reasons, which they are unwilling to trade off against other factors, and as such cannot be given a meaningful monetary value.² Similarly, some individuals may believe that permitting the loss of biodiversity is inherently wrong, or that humans have a duty to protect natural resources as stewards of the environment. Such intrinsic or ‘moral’ values can be contrasted with the instrumental values included in TEV estimation (see Figure 2 below). Instrumental values of biodiversity can be quantified according to the relative contribution that they make to human wellbeing. Changes in wellbeing may be expressed in monetary terms, by comparison with other goods and services that affect wellbeing and are traded in the market. Intrinsic values, however, cannot be compared with marketed goods and services because they do not have a quantifiable effect on human welfare.

As well as focusing on instrumental values, the concept of TEV is also limited to the extent that it considers only values as they relate to human beings. Philosophical debates persist regarding whether organisms or ecosystems can have intrinsic or instrumental values independently of the views or preferences of humans. However, these considerations will not be addressed here (see Kontoleon et al 2002). What is important for the purposes of this report is that such non-economic values can potentially be

² See Kontoleon et al. (2002) on the likely extent of such intrinsic values.

incorporated in decision making processes via alternative means but not via CBA. Methods for considering non-economic values include multi-criteria analysis as well as participatory and/or deliberative approaches. The latter have been suggested as complementary tools that can help overcome some of the limitations of monetary valuation, while allowing for individual preferences to inform environmental decisions (Kontoleon et al 2002).

1.4.5 Economic values included in case study analysis

The case studies reviewed in this report mainly provide monetary estimates of the use and non-use values of forest biodiversity, using the TEV framework, and on the values of marginal changes in biodiversity. As discussed above, it is not possible to obtain meaningful estimates of the total value of all forest biodiversity for a country or region, nor is it possible to express the intrinsic or ‘moral’ values of biodiversity in monetary terms. Furthermore, it would not be useful to do so for policy or other decision making purposes. The value of biodiversity in maintaining ecosystem resilience is also extremely difficult to value. However, some evidence is available from studies that attempt to estimate this insurance value, either theoretically or empirically, as discussed below.

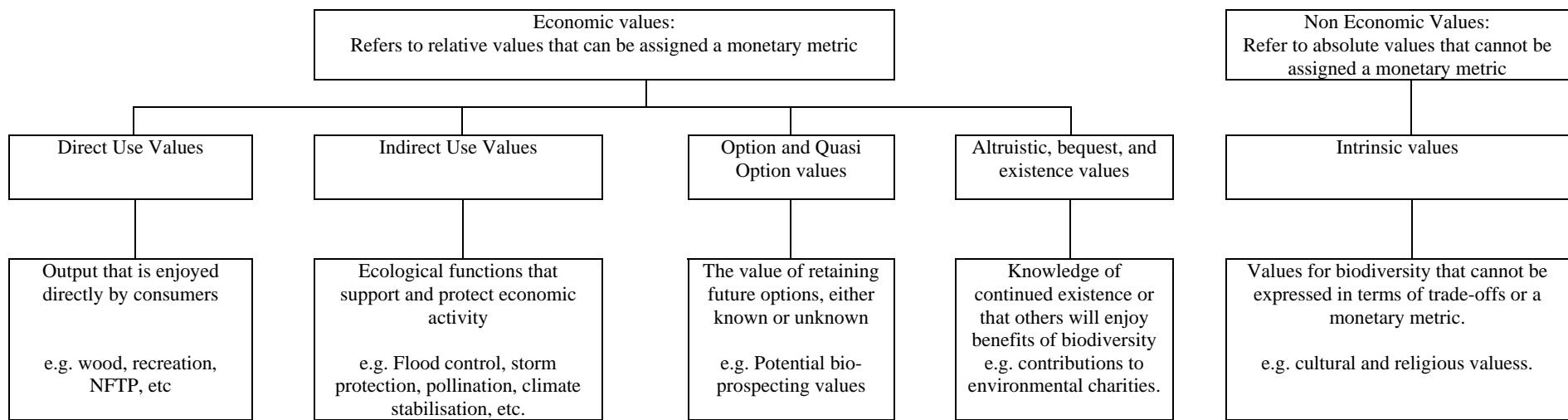


Figure 2 The different categories of value of biodiversity

Source: Adapted from Kontoleon et al 2002 and IIED 2003

1.5 The relationship between biodiversity and economic value

The typology of value discussed in the preceding sections is summarised in Figure 2. Forest biodiversity contributes to ecosystem services in two ways: first, it provides cultural services such as recreational opportunities or intangible spiritual, educational or aesthetic benefits; as well as provisioning services such as timber, food products or medicinal plants. Second, forest biodiversity maintains and enhances ecosystem functions, which in turn generate regulating services such as erosion control and climate regulation, and supporting services which are necessary for the production of all other ecosystem services. In addition to these impacts on ecosystem services, forest biodiversity increases the overall resilience of forest ecosystems, and increases the likelihood of all ecosystem services being maintained into the future.

Through the delivery of ecosystem services, forest biodiversity contributes in turn to the components of Total Economic Value as described in section 4.1: direct and indirect use values, and non-use values. As well as the contribution to TEV, the role of biodiversity in resilience has value as a form of insurance against the possibility of collapse in the event of external shocks.

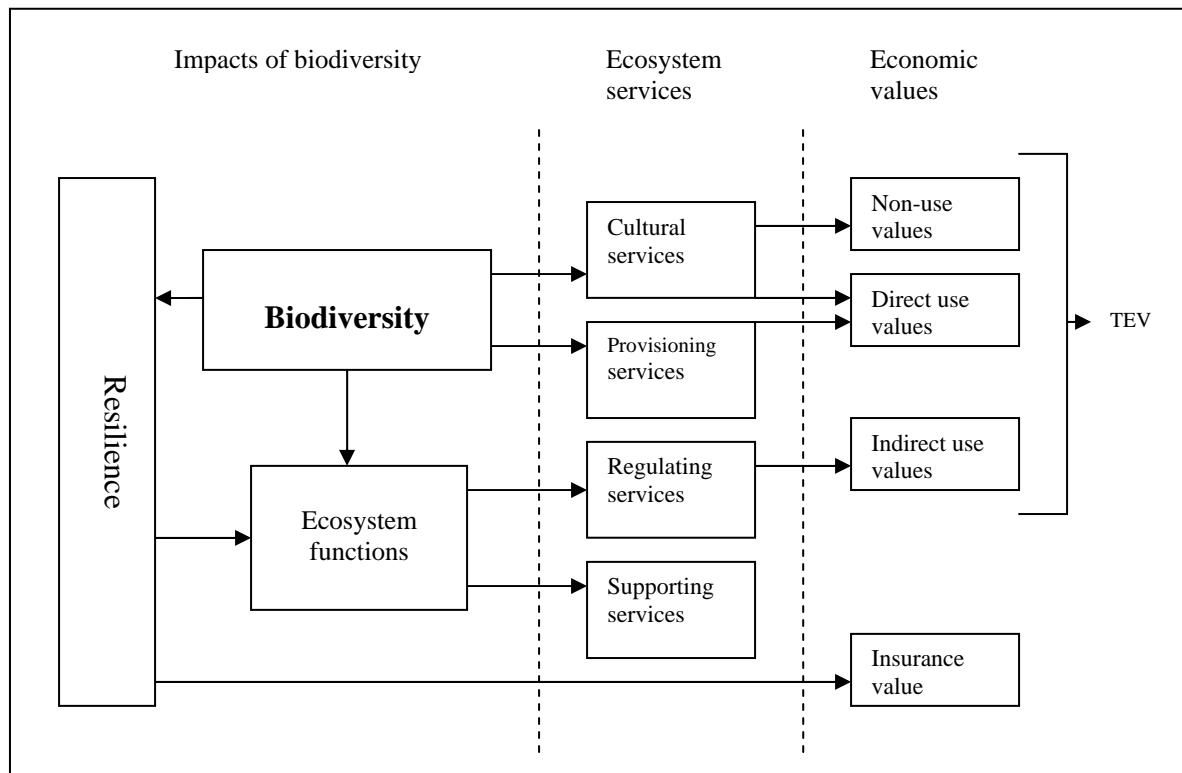


Figure 3 – The relationship between biodiversity and economic value

1.6 Welfare measures and total economic value.

The preceding section discussed which and in what manner the values of forest biodiversity and related ecosystem services can be meaningfully expressed in economic terms. The economic value of a change in the quantity or quality of an ecosystem service is referred to as consumer surplus. Four measures of consumer surplus are most relevant for the case of non-market goods, as shown in Table 1 below. These measures differ with respect to the reference level of welfare (utility) before the possible change takes place, the explicit or implicit property rights over the benefits arising from this change that are relevant in a specific context, as well as whether the change will, in fact, occur.

In a situation where the change involves an increase in utility (say to residents of Europe for the conservation of tropical forestland) the correct welfare measures, depending on the property rights assumed, would be willingness to pay (WTP) to obtain the change or willingness to accept payment (WTA) to forego it. In cases where the policy change entails a decrease in utility for the relevant population (e.g. rural communities in China subjected to a ban on forest use) the appropriate welfare measure would be WTP to avoid the change or WTA to tolerate it.

The difference (area) between the two demand curves, each corresponding to different levels of the non-market good, gives us a theoretically consistent estimate of these welfare measures. Obtaining monetary estimates of WTP or WTA requires access to some form of individual preference data for the non-market good or service in question. This is more easily said than done, as the data requirements and complexities of performing such an exercise are considerable. As discussed above, the very nature of non-market goods implies that their full value to society is not reflected in market prices and transactions. Economists have responded to this challenge by developing various non-market valuation tools. In the next section we briefly review the most commonly used valuation methods. This will help us comprehend the discussion in the next Chapter, which describes forest biodiversity value estimates obtained from a series of non-market valuation studies compiled for this report.

Table 1 Measures of welfare (economic value) for changes in forest policy

Welfare Measure	Initial level of good (reference level)	Proposed final level of the good	Proposed change in provision	Does the change in provision actually occur?	Actual final level of good	Reference level of utility	Property right to utility held by individual
WTP to secure a gain	Q0	Q1	Gain	Yes	Q1	U0	No
WTP to avoid a loss	Q1	Q0	Loss	No	Q1	U1	No
WTA to tolerate a loss	Q1	Q0	Loss	Yes	Q0	U1	Yes
WTA to forgo a gain	Q0	Q1	Gain	No	Q0	U0	Yes

1.7 Valuation Methods

Non-market valuation methods can be divided into formal valuation methods and environmental pricing techniques. Formal valuation techniques are further classified into revealed and stated preference techniques. Pricing techniques are classified into dose-response approaches, methods that rely on actual expenditures and methods that rely on potential expenditures for conserving biodiversity. The various empirical methods differ in the sources of the data that they use as well as the behavioural assumptions made with respect to the relationship between private goods and related non-market goods. What is important for the purposes of this report is to understand the main differences between these methods, their limitations, and which components of total economic value each method can potentially estimate. Figure 4 below shows how the various valuation approaches can be classified and how they relate to the total economic value of forest biodiversity.

1.7.1 Formal valuation techniques

Formal valuation techniques can be classified into revealed and stated preference methods.

Revealed preference valuation techniques rely on information from individual consumption/ purchasing behaviour occurring in markets over private goods that are related to the environmental resource in question. For example, housing demand data (obtained from markets for dwellings) can be used to infer the value of certain environmental services (e.g. the value of proximity to a diverse forest). Likewise, wage data (derived from labour markets) can be used to infer the value of workplace amenity or the value placed on environmental or health risks. In these two examples, the market for dwellings and the markets for labour act as ‘surrogate markets’ from which the value of the ‘non-market good’ can be inferred. To make this inference, a trained econometrician must obtain detailed information about the demand curve in the surrogate market used. In the examples used above, one must obtain information about the demand curve for ‘houses’ or for ‘labour’ in order to be able to make accurate inferences about the value of environmental benefits related to these goods. Because of this requirement of estimating demand functions, these techniques are also referred to as ‘demand curve’ valuation techniques.

Commonly used revealed preference valuation methods include:

- Travel cost methods: these are mostly relevant for determining the recreational values associated with biodiversity. They are based on the rationale that recreational experiences are associated with a cost (direct travel expenses and the opportunity costs of time). The value of a change in the quality or quantity of a recreational site (resulting from changes in biodiversity) can thus be inferred by estimating the demand function for visiting the specific site.

- Hedonic pricing: Houses or property in general consists of several attributes, some of which are environmental in nature (e.g. proximity of a house to a forest, good soil retention qualities in a plot of land, clean air etc.). Hence, the value of a change in biodiversity will be reflected in a change in the value of property (either built or land). By estimating a demand function for property, the analyst can infer the value of a change in such non-market environmental benefits.
- Wage differential approaches: These methods are useful for assessing the value of environmental amenities from data on wage rates. However, as labour markets in some developing countries are incomplete, information on wages is often suspect and this technique might be impractical.

Revealed preference methods have the appeal of relying on actual/observed behaviour. Their main drawbacks are the inability to estimate non-use values and the dependence of the estimated values on the technical assumptions made with respect to the relationship between the environmental good and the surrogate market good.

Stated preference valuation techniques are used in situations where both use and non-values are to be estimated and/or when no surrogate market exists from which environmental (use) value can be deduced. These techniques use questionnaires to develop a hypothetical market through which they elicit values (both use and non-use) from survey respondents for the environmental good under investigation. Stated preference techniques do not suffer from the same technical limitations as revealed preference approaches and can also be applied to non-use values. However, the hypothetical nature of the market constructed has raised numerous questions regarding the validity of the estimates.

The main types of stated preference techniques are Contingent valuation (CV) and Choice Experiments (CE). The main difference between the two approaches is that CV typically presents respondents with one option that is associated with some price (varying across respondents). Respondents are asked to vote on whether they would be willing to support this option and pay the price or if they would support the status quo (and not pay the extra price).³ In the case of CE, survey respondents are given a choice between several policies, each consisting of various attributes one of which is either a price or subsidy. One of the alternatives offered is the status quo option. Respondents are then asked to consider all the policy options by balancing (trading off) the various attributes.

Either of these techniques can be used to assess the total economic value resulting from a change in the quantity and/or quality of an environmental resource. Though the CV method is less complicated to design and implement, the CE approach is more capable of providing value estimates for changes in specific characteristics (or attributes) of an environmental resource.

³ If a WTA scenario is involved a policy option is described to respondents that is associated with a specific subsidy amount. Respondents have to decide if they would want to support the policy and receive the subsidy or support the status quo and not receive any subsidy.

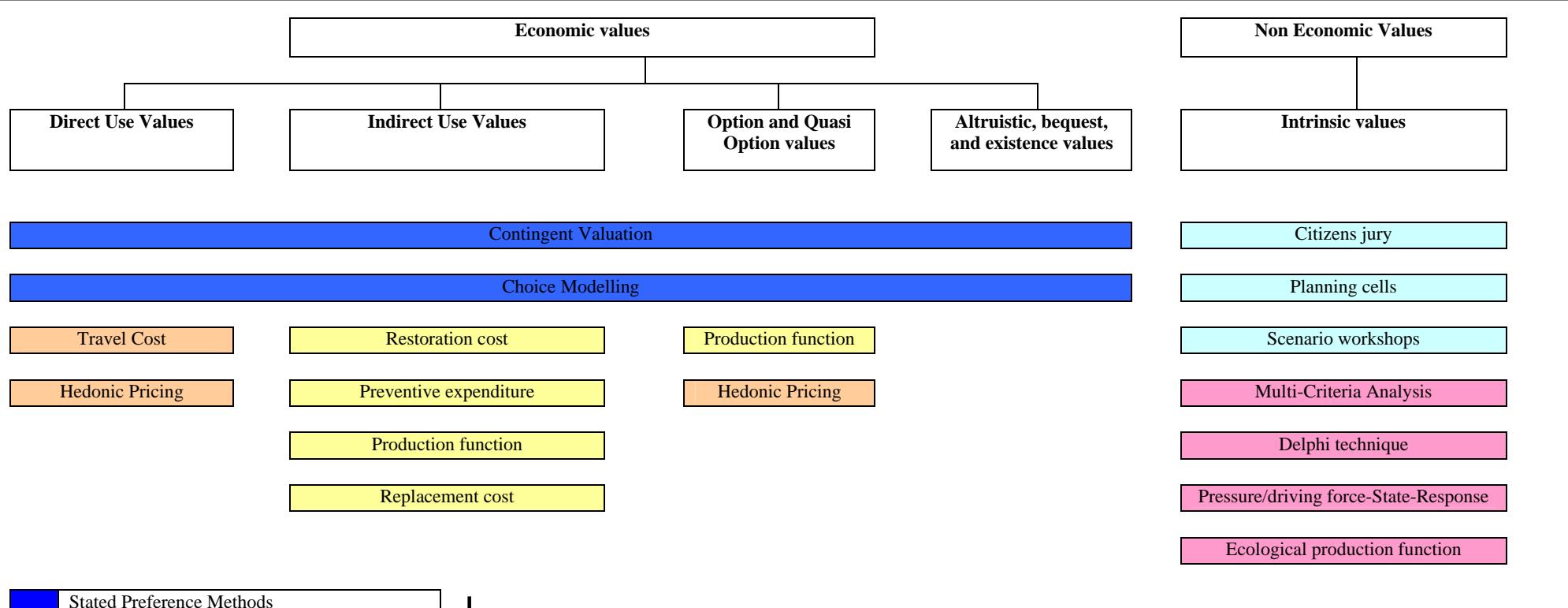


Figure 4 Categories of values and corresponding valuation methods
 Source: Adapted from Kontoleon et al 2002 and IIED 2003

1.7.2 Environmental pricing techniques

Environmental pricing techniques rely on available market price and output information to determine the economic value of environmental goods and services. They can be divided into three main categories:

a) Methods in which market prices are used to value the impacts of a change in biodiversity on the productivity and output of a marketed good or service. These methods are also referred to as 'Dose-Response Approaches'. In these methods, the quality or quantity of an environmental resource is treated as an input to the production of one or more marketed goods and services (outputs). Changes in these environmental inputs may lead to changes in productivity or production costs which, in turn may lead to changes in prices and output levels which can be observed and quantified (Dixon, et al., 1994). Examples of such methods include:

- Changes-in-productivity approaches: These involve identifying the changes in productivity resulting from a change in biodiversity. For example, changes in grassland biodiversity may be reflected in the value of milk and dung produced by local farmers.
- Loss of earnings approaches: here the value of a change in biodiversity is reflected in change in human labour productivity. The analyst can use available data on wages or health expenditure to infer the value of lost earnings or increased medical costs.
- Opportunity cost approaches: These measure the value or 'cost' of conserving biodiversity in terms of the benefits that must be foregone by doing so.

As in the case of revealed preference methods, discussed above, dose-response approaches collect information from surrogate markets that are directly related to the environmental good. However, these pricing methods differ from demand curve methods in that the analysis need not estimate the entire demand curve for the surrogate good. Instead the analyst can infer some measure of biodiversity value *directly* from price and output data from the surrogate market.

b) Methods in which market prices are used to value the costs actually arising from a change in biodiversity. The second set of pricing techniques relies on data from actual costs of maintaining or preventing environmental degradation as a proxy for environmental value. This set of valuation methods includes:

- Cost-effectiveness analysis: where a predetermined objective with respect to the quantity or quality of an environmental resource is established and then the most cost-effective means of achieving it are identified and valued.
- Preventive or mitigation expenditure approaches: where the value of an environmental resource is approximated by the cost of preventive measures that people are willing to pay to avoid damaging it, or by the savings obtained from a reduction in maintenance costs due to avoided damage.

c) Methods in which market prices are used to value the costs potentially arising from a change in biodiversity. The third set of pricing methods is similar to (b) above but relies on potential (as opposed to actual) costs as proxies for environmental value. These include methods as such as replacement cost or relocation cost approaches and shadow-project appraisal.

1.7.3 Using forest biodiversity valuation case studies

The discussion so far has focused on how the impacts of changes in forest biodiversity can be quantified in economic terms. In general, this is done by analysing the influence of biodiversity on the quantity or quality of services provided by intact, well functioning, ecosystems, and the contribution of those services to human welfare. However, there are some additional issues to consider when using the results of case studies to assess the economic value of forest biodiversity.

The first issue is the relatively small number of case studies that directly value the benefits of biodiversity, compared to the much larger number of cases that value a particular biological resource. As noted above, biodiversity contributes to the value of specific ecosystems or biological resources, and on this basis we include studies that estimate the ultimate value of those contributions. However, we also seek to highlight studies that directly value the benefits of biodiversity itself.

Another issue relates to the distinction between economic stocks and flows. So far we have focused on the value of flows of goods and services that are provided by a certain stock of biodiversity. It is also possible to assign value to the stock itself, which would represent the total (present) value of actual or potential flows over some time period. Both of these are valid and should be equivalent, but when looking at values estimated in case studies it is important to be clear about whether stocks or flows are being valued.

Finally, there is the question of who receives or experiences the benefits of forest biodiversity. Some benefits will be local in nature, such that people in the immediate vicinity gain the most, while values decline with distance from the resource. Other benefits, such as existence value or carbon sequestration, are global in nature and may be just as valuable to people far from the resource as to those with direct access to it. This distinction is important when aggregating estimates of values from study samples to the relevant populations, or transferring benefits from one site to another. This last issue is considered in more detail below.

1.8 Benefit transfer methods

All of the techniques discussed above involve significant estimation complexities and data collection and processing requirements that require highly specialised expertise as well as considerable time and money. In many situations, it may not be feasible to obtain the required specialist expertise or necessary funding to undertake original data collection and analysis. As an alternative, under certain circumstances, the analyst may employ techniques that utilise estimates of forest values obtained in one context as proxies for forest values in another context. Such techniques are referred to as 'benefit transfer' (BT)

methods. The site or source where the original valuation estimates come from is usually referred to as the ‘study site’, while the location where the data are transferred is called the ‘policy site’ (see Navrud and Ready 2007).

1.8.1 Requirements for accurate benefits transfer

All individual valuation studies rely on statistical methods and thus they inevitably entail some degree of error (due to various sources such as measurement and specification). BT unavoidably adds a further layer of statistical error, due to differences between the study site and the policy site. Although an original valuation study is always the ‘first best’ option, benefit transfer is an acceptable ‘second best’ strategy when faced with situations where: (i) budget and time constraints are binding; (ii) the environmental impacts being examined are likely to be low and do not justify the time and costs of an original study and (iii) a high degree of accuracy is not required. BT can be performed with a reasonable or at least acceptable degree of accuracy when the following three main conditions are met (see Desvouges et al. 1998):

- 1) The policy site should be thoroughly described, including:
 - Extent and magnitude of the policy site or expected resource changes;
 - Size and characteristics of the population that will be affected by the policy site and/or changes to it;
 - Data needs for an economic assessment, including the type of measure (unit, average or marginal value), the values being considered (use, non-use, total value), and the degree of accuracy and precision required for the transferred data.
- 2) The study site data should likewise satisfy certain conditions for use in benefit transfers, including:
 - Value estimates transferred must be based on adequate data, sound economic method, and correct empirical technique
 - The original valuation study should contain detailed information on the statistical relationship between benefits (costs) and (a) the socioeconomic characteristics of the affected population and (b) the physical/environmental characteristics of the study site.
 - An adequate number of individual studies for similar sites should ideally be available in order to enable credible statistical inferences concerning the applicability of the transferred value(s) to the policy site.
- 3) Finally, the correspondence between the study site and the policy site should exhibit the following characteristics:
 - The environmental resource and the change in the quality (or quantity) of the resource at the study site should be similar to the resource and expected change at the policy site. This similarity should include the nature of the change and possibly also the source of that change.
 - The institutional settings between the study and policy site should be similar. There should also be similarity with respect to demographic and cultural variables.

1.8.2 Benefit Transfer Approaches

There are two main approaches to benefit transfer, the ‘unit value’ and the ‘function transfer’ methods.

Unit value transfer

This approach involves directly transferring the (mean) benefit estimate (e.g. mean WTP/household/year) from the study site to the policy site. It assumes that the change in well-being experienced by an average individual at the study site is the same as that which will be experienced by the average individual at the policy site. The values being transferred may derive from a single study (point value transfer) or from several related studies (average value transfer)

Simple unit transfer is normally not considered appropriate when values are transferred across countries/regions with varying income levels. Transferred values would then require some sort of income adjustment (for example, using purchasing power parity indexes). However, countries/regions differ in other ways besides income levels (e.g. preferences, institutions etc) and simple income adjustment may not be sufficient to capture these differences.

Benefit function transfer

Instead of transferring benefit estimates (i.e. values), the analyst may transfer the entire benefit or demand function. The main advantage of transferring an entire demand function to a policy site is the increased precision of tailoring a benefit measure to fit the characteristics of the policy site. The benefit relationship to be transferred from the study site(s) to the policy site could again be estimated using either revealed or stated preference approaches. For example, from a generic CV study we could derive a benefit function of the following simple form:

$$WTP_i = b_0 + b_1 G_j + b_2 C_i \quad (1)$$

Where:

WTP_i = WTP for household i

G_j = the characteristics of the environmental good j

C_i = the characteristics of the household i

To implement this approach the analyst would need to:

- a) Identify a study from the literature which can provide reliable estimates of the parameters (b_0 , b_1 , and b_2) for an appropriate study site.
- b) Collect data on the same independent variables (G_j and C_i) at the policy site (these may be available from national statistics).

- c) Plug in the values of G_j and C_i from the policy site using the original WTP function.
- d) Estimate predicted WTP (i.e. expected WTP or $E(WTP)$).

Meta-Analysis is another form of benefit function transfer. Generally speaking, meta-analysis involves statistical analysis of data from a large number of case studies. Results from each study are treated as a single observation in an analysis of the combined data set. Meta-analytic techniques can be used to derive a single WTP value, following the logic of average value transfer. Alternatively, meta-analytic techniques may be used to estimate a general benefit function.

Any meta-analysis based on the data compiled for this report should be undertaken with caution. This is because the data comes from studies that estimate different values, using different techniques and focusing on different geographic scales and time horizons. More reliable results would be obtained by focusing on 'comparable' studies (e.g. a meta-analysis of forest eco-tourism values).

1.8.3 Benefit Transfer – Guidance for policy makers

There are three main difficulties or challenges in benefit transfer. These are:

- 1) Availability and quality of existing studies. Benefit transfers can only be as accurate as the initial value estimates. Also, unit value estimates can quickly become outdated.
- 2) Valuation of new policies or projects may be difficult on account of:
 - the expected change resulting from a policy is outside the range of previous experience;
 - The study site relates to large changes in the non-market good while the policy site involves marginal changes (or vice versa); or
 - The study site relates to an improvement in the quality or quantity of the non-market good while the policy site involves a decrease (or vice versa)
- 3) Differences between the study site(s) and policy sites that are not accounted for in the specification of the valuation model or in the procedure used to adjust the unit value.

Efforts to address these issues aim at reducing the so called 'transfer error'⁴. Complex (and expensive) methodological work is currently underway to find ways to achieve acceptable levels of 'transfer error' (estimated around 20-30%).

⁴ Formally, the transfer error of a study is given by the following formula:

$$\text{Transfer Error} = \frac{|\text{Transferred estimate} - \text{Target Country Estimate}|}{\text{Target Country Estimate}}$$

The transfer error can only be reliably assessed if original valuation work is undertaken in both the study and policy site. See for example, Ready et al 2004 , Brander (2004) and Shrestha and Loomis (2001).

The general response to the challenges of benefit transfer is twofold: first, develop a protocol for benefit transfer and, second, establish sufficient, up-to-date and consistent non-market valuation databases. There have been recent advances in both areas. For example, based on a review of value transfer studies and validity tests of transfer, Brouwer (2000) propose the following seven-step protocol for good practice when benefit transfer is used in CBAs:

1. Defining environmental goods and services
2. Identifying stakeholders
3. Identifying values held by different stakeholder groups
4. Stakeholder involvement in determining the validity of monetary valuation
5. Study selection
6. Accounting for methodological value elicitation effects
7. Stakeholder involvement in value aggregation

EEA (2007) highlights the particular challenge of transferring calculated values from a study site to a policy site that is much larger (or smaller) in geographic scale. This issue may be particularly relevant in the case of forest values, due to their geographic scope. Some researchers suggest that this aggregation challenge may be addressed by adopting a spatially-explicit benefit transfer approach, using recent advances in GIS mapping techniques (see Troy and Wilson 2006).

Finally, several web sites containing useful information on BT have been established in recent years, including (McComb et al. 2006):

- EVRI - Environmental Valuation Reference Inventory
<http://www.evri.ca/>
- ENVALUE environmental valuation database:
<http://www.epa.nsw.gov.au/envalue/>
- Valuation Study Database for Environmental Change:
<http://www.beijer.kva.se/valuebase.htm>
- The New Zealand Non-Market Valuation DataBase:
<http://learn.lincoln.ac.nz/markval/>
- RED Data Base: <http://www.red-externalities.net/>

1.9 Using valuation methods in forest policy assessment

Having summarised the main valuation tools available for estimating forest biodiversity values we now turn to some practical considerations when using these tools in forest policy assessment.

The choice of which valuation method(s) to use for assessing forest policies depends largely on the conditions within which such an exercise is to be performed. Ideally, where time and financial constraints and access to specialised expertise do not pose problems,

the use of demand side approaches (stated and revealed preference studies) may be more appropriate. This is because:

- i) Demand side valuation approaches (and in particular stated preference studies) are more flexible and better able to derive the full set of economic values for multi-faceted resources such as forest biodiversity (Randall 2002). Pricing methods may however be used to test the validity of the values derived from demand-side valuation.
- ii) While the amount of information needed to undertake a reliable stated preference study may be significant, this information is often more accessible (compared to that required for a revealed preference or pricing method) as it is based on eliciting peoples' preferences directly using standardized survey methods.
- iii) A further advantage of stated and revealed preference valuation approaches is their potential multi- or inter-disciplinarity. This stems from the fact that such methods can include the values and perceptions of a wide range of stakeholder (including indirect beneficiaries at the global level). Moreover, such approaches are readily integrated with participatory methods.
- iv) Finally, stated preference methods are best able to assess the full range of economic values, including various forms of use as well as non-use values. Such an approach yields more holistic estimates of value and avoids the problems of double-counting which may arise from independent, piece-meal valuation (Randall 2002). It should be noted, however, that even stated preference methods may under-estimate true economic value. This is because people typically overvalue small things and undervalue larger ones.

When a holistic, original, demand-side valuation study is not feasible, a policy maker may select an appropriate valuation method from the decision tree shown in

Figure 5, below. As a first step, the analyst should assess whether the policy under consideration is likely to have a significant impact on forest biodiversity. If the answer is no, then it may be possible to avoid the cost and effort required for an original valuation exercise in favour of benefit transfer. Two alternative pathways (denoted B1 and B2 in

Figure 5) can be followed, depending on the availability of previous valuation studies.

If the answer to the first question is positive, then obtaining an estimate of these significant impacts on forest biodiversity would be warranted. Where time and budget allows, one may opt for a stated preference study (such as a CV study) to derive a holistic, multiple-output value of forest biodiversity at an appropriate geographic scale (e.g. global, continental, or local).

Apart from contingent valuation, a contingent choice experiment or the conjoint technique can also be used to estimate local, demand-based values of biodiversity. Using the latter techniques, people express how they compare and perceive the potential for

substituting their demand for non-marketed biodiversity good(s) and at least one marketed good (e.g. NFTP). If the conjoint technique is used, different levels of biodiversity-related goods need to be balanced in the demanders' perceptions.

Where time and budget constraints are limiting, policy makers may consider whether secondary surrogate markets exist in which to undertake revealed preference or pricing studies, such as a hedonic or travel cost analysis, or replacement cost and averting behaviour analysis. The choice of method(s) will depend on the data available and the values that the analyst wants to focus on. If appropriate data is not available, the analyst has no other choice but to resort to benefit transfer techniques to obtain some measure of the economic value of forest biodiversity. While the estimates obtained from benefit transfer may not be entirely accurate, they can still be considered in any policy assessment as the alternative (i.e. not accounting for the value of biodiversity loss) is likely to be even less precise!

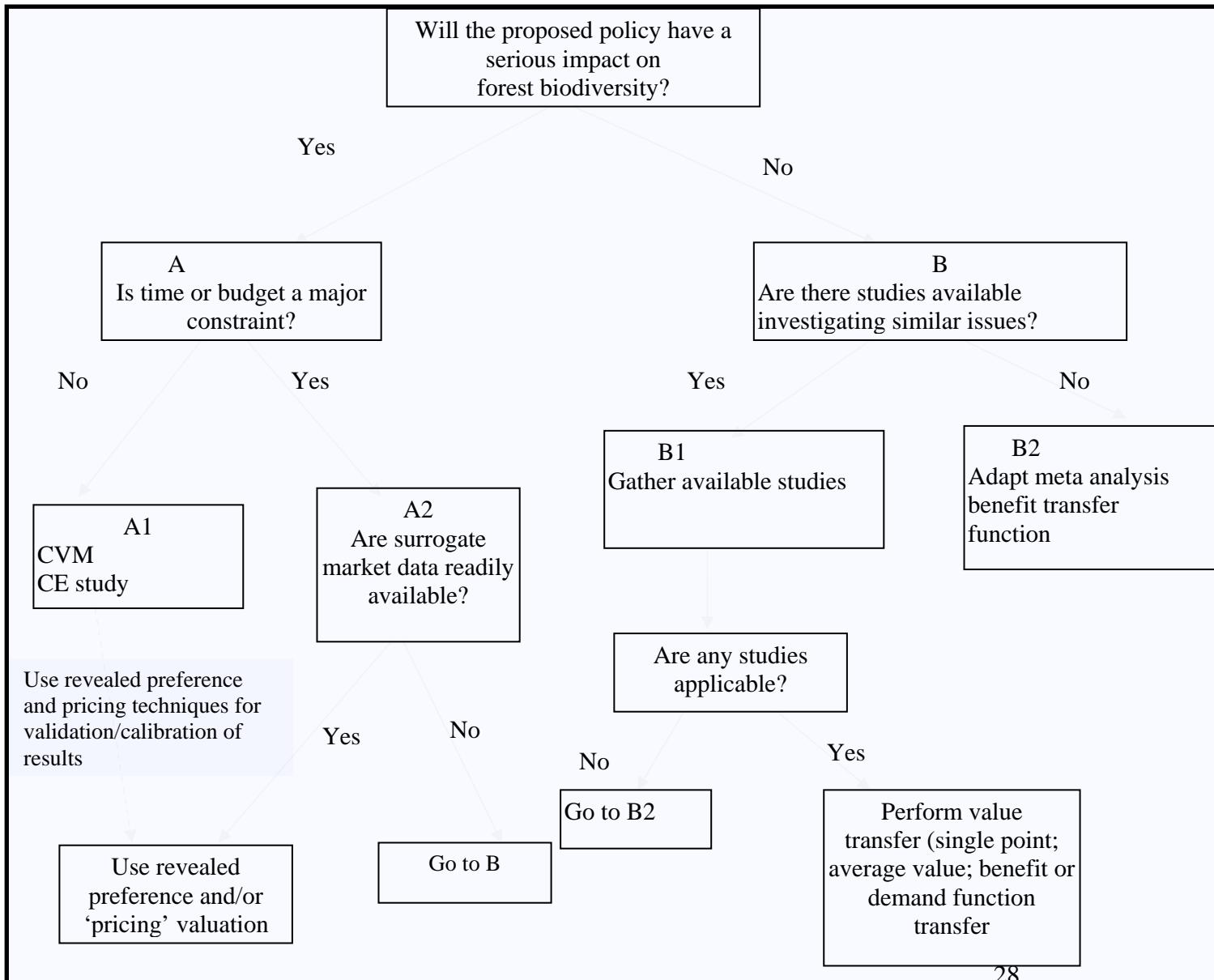


Figure 5 Decision tree for choosing valuation methods for integrated assessment

Source: Adapted from Navrud and Ready (2007)

1.10 Costs of biodiversity conservation

In addition to estimating the economic benefits of conserving forest biodiversity, using the methods outlined above, it is often helpful to compare these benefits with the costs of conservation. The main reason for doing so is to help ensure the efficient use of scarce funds, by focusing conservation efforts where the net benefit is greatest. Another important reason is because conservation costs may not be borne equally by all stakeholders; without comparable information on benefits and costs, it can be difficult to achieve equity in burden sharing or benefit distribution.

1.10.1 Opportunity costs

The most significant costs of conserving forest biodiversity are often the opportunity costs of retaining land in a more-or-less natural state, rather than using it intensively or converting it to some use that is incompatible with biodiversity conservation (e.g. a parking lot or industrial facility). Forest land may be converted to agriculture, used for urban development, or managed in order to increase the output of timber or another valuable forest product. In all of these cases, some components of biodiversity may be lost, but other benefits will be obtained. The benefits may include food or cash income for farmers, employment opportunities for local households, or profit for timber companies. If these opportunities are not accounted for, the costs of losing biodiversity, or the benefits of conserving it, will be overstated.

Another reason for accounting for opportunity costs is to avoid double-counting the benefits of biodiversity. As discussed above, biodiversity and biological resources provide many different services with market and non-market values. However, these services may not all be complementary (Turner et al, 2003). For example, extracting timber from a forest may reduce its value for recreation. Alternatively, within the category of recreation, improving access or facilities in a forest may increase the benefits obtained by some users, but reduce the value for others who would prefer a less disturbed natural environment.

Case studies that measure opportunity costs do so in a number of ways. The most straightforward approach is to use market prices of comparable land as a measure of the highest valued alternative use of a forested area. If all values of alternative uses are adequately reflected in land prices, then these can be directly compared with the market and non-market benefits of maintaining the biodiversity of forest land. However, land

prices may not accurately reflect all values of alternative uses. For example, agricultural subsidies may increase the market returns to agriculture above the real social benefits of agriculture. Conversely, planning restrictions on how land can be used may reduce its market value. In addition, some countries or regions (particularly in the developing world) may not have well-functioning land markets, with the result that land prices are not available or are too scarce to be reliable.

Other measures of opportunity costs may be based on estimates of the returns to households or firms from particular uses of land. In some cases this is done by modelling the productivity of the land and the expected market value of output. Other studies survey households or firms, following the implementation of a conservation programme, to assess changes in income from forest land use. Clearly these methods will only provide valid estimates of opportunity costs if they account for the costs of the alternative activity as well as the income that could potentially be earned, including both the costs of inputs to production and the costs of converting the land.

1.10.2 Implementation costs

As well as the opportunity costs of conserving forest biodiversity, there will also be costs associated with implementing any conservation programme or policy. Implementation costs will be incurred at all stages, from gathering information about what to conserve and what methods to use, to managing the implementation process, enforcing any restrictions, and monitoring the programme's success. Studies that quantify these costs generally do so by examining expenditure on different elements of existing conservation programmes.

1.10.3 Costs of alternative policy mechanisms

Both the magnitude and distribution of costs will vary according to the policy instrument or mechanism that is used to achieve a given conservation outcome. A key aspect of the distribution of costs is who incurs the opportunity costs. A programme in which farm households are paid a subsidy for not converting forest land will place the burden of the opportunity costs on whoever is funding the conservation programme, whether that is the national government, a multilateral organisation, or a private company. In contrast, the costs of a traditional protected area programme may appear lower on paper because the funding body does not cover opportunity costs, while households who would otherwise convert the land or use it for resource extraction bear the costs instead.

The distribution of costs can also affect the magnitude of implementation expenditure. For example, a protected area programme that restricts the formal or informal rights of households or firms to use forest land, without compensation, may incur relatively high enforcement costs. Alternatively, a programme involving payments to individual farmers for environmental services would be expected to have lower enforcement costs, but higher transactions costs from administering large numbers of individual contracts.

Variation in the absolute costs of alternative programmes cannot be looked at in isolation, but must be related to the benefits achieved by those programmes. A conservation programme involving detailed targeting of particular biodiversity objectives may be more costly to implement than one that simply sets aside areas of marginal land. However, the question of interest is the relative efficiency or cost effectiveness of the programmes, in terms of maximising the return to conservation spending. The case study analysis presented below aims to compare evidence on the costs and benefits of conserving forest biodiversity in different locations, different types of forest, and using different policy mechanisms.

1.11 Comparing costs and benefits

Having discussed some conceptual and empirical issues in assessing the benefits and costs of forest biodiversity, we now turn to how estimates of benefits and costs can be analyzed in a structured way, in order to determine the net benefit of a conservation programme. The structured comparison of costs and benefits for policy appraisal typically involves the following major steps:

- 1) Specify the set of policies or projects that are to be compared.
- 2) Specify that changes to forest biodiversity that will result from these policies or projects, as well as the type of benefits and costs that will be affected.
- 3) Choose the type of welfare measures that need to be evaluated (e.g. WTP, WTA).
- 4) Decide whose benefits and costs should be counted (i.e. what scale should we aggregate over or who has standing, from local to global levels).
- 5) Predict the impacts quantitatively over the life of the project or policy.
- 6) Monetize all impacts (benefits and costs).
- 7) Discount benefits and costs occurring at different points in time, in order to obtain present values.
- 8) Compute the net present value of each alternative policy or project.
- 9) Perform sensitivity analysis on key variables.
- 10) Make a recommendation based on the result.

Each of these steps entails various challenges. Extensive discussions of the complexities involved can be found in Broadman et al (2006), Brent (1996) and Layard and Glaister (1994). In this section, we selectively review some key issues that have been identified as especially relevant for the purposes of this report.

1.11.1 CBA and discounting

As biodiversity benefits and costs accrue over time, their valuation involves a temporal dimension. An important step in CBA is therefore to determine how to compare costs and benefits that arise at different points in time. The simplest approach is to assign an equal weight to all values across all time periods. This amounts to setting the 'discount rate' at zero. However, such an option would be descriptively inaccurate, as people do in fact discount future benefits and costs due to their 'time preference'.

Ignoring the reality of positive time preference is analogous to ignoring society's preferences concerning the environment. Moreover, we might expect that future generations will be better off than people living today, due to technical progress and general economic development. Again, this would imply assigning a positive discount rate to future costs and benefits, assuming a declining marginal utility of consumption as incomes rise over time.

Choosing a low discount rate (or setting it at zero) implies that future consumption matters more and present consumption matters less, and thus more savings (i.e. investment) should take place in the present. If the discount rate is zero, then presumably people should save all of their income. In short, low or zero discounting implies large sacrifices of current well-being, with ethical implications that few would find acceptable (Pearce et al 2003).

At the same time, the logic of a positive discount rate must be balanced against the equity requirements of sustainable development. Following the latest developments in discounting research (e.g. Weitzman 2001 and Gollier 2002), which attempt to address this challenge, policy makers could adopt a declining or hyperbolic (as opposed to exponential) discount rate. There are several reasons to support such a time-declining social discount rate (Boardman et al. 2006):

- 1) Empirical evidence suggests that people use lower discount rates for events that occur farther into the future.
- 2) Long-term environmental consequences have very small present values when discounted using a constant rate, implying that spending a relatively small amount today to avert a costly disaster several centuries in the future is not cost-beneficial.
- 3) Constant rates do not appropriately take into account the preferences of future, as yet unborn, generations.
- 4) Constant rates do not appropriately allow for uncertainty as to market discount rates in the future. Allowing for this uncertainty implies that lower discount rates should be used to discount consumption flows that occur farther in the future.
- 5) Declining discount rates are also consistent with a risk premium for potential non-marginal changes in the flows of ecosystem goods and services in the far future.

As forest policies are inherently long run policies, due to slow tree growth and long rotation periods, the arguments summarize above for a time declining discount rate are of particular relevance. In practice, one strategy is to use a constant discount rate for the first 30-50 years of a project time horizon (say between 5-8% to reflect current social time preference), together with a reduced or gradually declining rate thereafter (between zero and 1%). Such an approach may seem clumsy but it has empirical support and would avoid short-changing either the present or future generations.

1.11.2 CBA and the distribution of wealth

Any comparison of the costs and benefits of alternative uses of forest land, as implied by the standard CBA utilitarian framework, ultimately depends on the distribution of wealth at a given point in time. This is because prices and values reflect preferences, which vary in part depending on peoples' circumstances and relative well-being. In short, if the distribution of wealth were to change, then the net benefits of a conservation policy may also change.

The link between net benefits and the distribution of wealth may not be a problem, if the losers from a policy are compensated (as required by the simple Pareto principle). However, as discussed above, CBA operationalises the *potential* compensation Pareto principle, under which it is conceivable that an apparently efficient policy could *reduce* aggregate social welfare, if individuals with different levels of wealth have different marginal utilities of money. In other words, the potential compensation Pareto principle may be undermined when costs and benefits are unevenly distributed across different wealth or income strata. Nevertheless, some economists argue that if the potential Pareto principle were applied consistently, net winners and net losers would tend to even out and the overall effect would be an increase in utility for everyone (Broadman et al., 2006).

Critics of CBA continue to question the validity of Pareto efficiency as a decision making tool, as it depends on the present distribution of income. This issue is of particular relevance when comparing costs and benefits of forest biodiversity, as in many cases those who bear the costs of conservation (or the opportunity cost of non-conversion) are amongst the poorest income groups and at the same time represent a significant portion (if not the majority) of the people who should be counted. This disparity is often observed when comparing costs and benefits of forest policies *within* developing countries but is also relevant when undertaking such comparisons at the international or global level. For example, recent efforts to conserve natural forests in China, in order to address regional environmental problems (such as flooding) appear to impose disproportionate costs on the rural poor. Similarly, the opportunity costs of conserving the Amazon forest (which provides a global public good benefit) are likely to fall disproportionately on relatively poor rural communities.

From an analytical perspective, one solution to this problem is to use some form of *distribution weights* when estimating net benefits, especially in situations where there are large disparities in the distribution of income and/or when costs disproportionately fall on the lower income strata of the populations being considered. Such an approach involves estimating net benefits separately for each of several groups, which may be distinguished by wealth or some other social criterion. The net benefits for each group are then multiplied by a weighting factor and summed to determine the (socially weighted) net benefit of the policy or project. The main challenge in such an approach is choosing an appropriate set of weights, such as a weight inversely proportional to wealth (or income) or placing a higher weight on those with wealth below a certain threshold (e.g. a predetermined poverty level) (Broadman et al, 2006).

1.11.3 CBA and standing

The issue of whether and how temporal and social disparities should be addressed in CBA was discussed in the previous two sections. In the case of discounting, we found support for a decreasing weight (at a declining rate) of future preferences, while in the case of distributional impacts we observed that socially-disaggregated and weighted CBA could be performed. Another important issue that arises when comparing costs and benefits in policy appraisal is whose preferences should be counted or, in other words, who has ‘standing’. The issue of standing raises at least two key challenges (Kontoleon et al, 2002).

First, the geographical boundaries (or scale) of the CBA calculation need to be determined. Undertaking CBA within a particular country does not normally pose problems with respect to standing. However, in cases where policies impose costs or generate benefits that extend across national boundaries (as is the often case with forest values), the question of standing is not so clear cut. Similar problems may arise at the national level, where a government wishes to examine the impacts of a policy at the provincial or county level. To address this problem, the analyst can conduct parallel analyses at different levels (i.e. local - national - global) (Broadman et al, 2006).

Second, the question of whose preferences count *within* a geographic boundary needs to be addressed. For example, should the preferences of illegal aliens, citizens living abroad or legal foreign residents be counted? An obvious solution would be to confine CBA only to those who have legally defined rights. However, this may raise other problems, as many societies include people with real preferences and economic clout but limited legal rights, for example women in some countries. Basing an analysis of net benefits on the basis of who has legal rights may not be justified on moral grounds. Some people extend this principle to argue that the ‘preferences’ of non-humans (plants and animals) should also count in CBA. Such an approach is incompatible with the CBA framework, which can only handle anthropocentric values, although the preferences of non-humans may be indirectly reflected in the consideration of non-use values, which may be motivated in part by a form of altruism towards non-human species.

Yet another approach is to view standing as a matter of degree. In this case, one would assign to each individual some kind of weight in the aggregation process but this would *not* be done on equity grounds, as discussed above, but rather on the basis of geographical proximity and/or familiarity with the good under investigation. Many forest valuation studies are undertaken among populations living near a particular forest ecosystem. Using average values obtained from such studies as representative of an entire country, for example, may not be appropriate. In this case, using some form of distance-decay formula may be recommended (e.g. Bateman, 1999). Alternatively, when considering the non-use values associated with forest ecosystems, the question is whether individuals having no prior knowledge of a particular ecosystem should be granted full (if any) standing. For example, Dunford et al. (1997) and Johnson et al. (2001) have argued that demand for knowledge about the resource and/or its injury are required for a person’s non-use values to have legal standing. Where this is not the case, it may be appropriate to

discount the preferences of ignorant or unconcerned individuals in the aggregation process (see also Randall 1997, Zerbe 1991).

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CHAPTER 2

CASE STUDY EVIDENCE OF THE BENEFITS AND COSTS OF CONSERVING FOREST BIODIVERSITY

CHAPTER 2

THE BENEFITS AND COSTS OF CONSERVING FOREST BIODIVERSITY: CASE STUDY EVIDENCE

2.1 Introduction

In this chapter we look at the findings from more than 200 case studies⁵, which estimate the benefits and costs of conserving forest biodiversity. The primary criterion for selecting studies was the extent to which they focus on the *diversity of forest resources*, as opposed to other types of biodiversity or forests in general. Many of the studies estimate the costs or benefits of conserving forests as a whole, or the costs of conserving biodiversity across all ecosystems. In fact, relatively few studies focus specifically on forest biodiversity. Studies focusing on forests with relatively low levels of diversity, such as those managed for optimal timber growth, have been excluded. All of the studies selected provide some information, directly or indirectly, on the values of diverse or unique forest types, or the costs of maintaining or enhancing forest diversity.

The case studies considered here come from various sources, including scientific journals, reports to governments and conservation agencies, and unpublished working papers. Preference was given to peer-reviewed sources; studies which lacked a clear and consistent methodology for estimating values were not included. However, we do not attempt to compare the quality of individual estimates, beyond highlighting where values are obtained from meta-analysis of multiple studies.

Note that the sources from which the case studies are drawn (mainly journals, public agencies and private research institutes) may be subject to some degree of implicit or explicit censoring, in that they may more frequently contain studies that indicate a cost-benefit ratio above unity. Overcoming the bias caused by this form of censoring is not simple, although one can try to assess its magnitude. Pearce (2007) compares biodiversity values derived from non-market valuation studies with actual expenditures on biodiversity conservation and finds a large disparity, with actual expenditures falling considerably short of estimated values. Such divergence is to be expected, in part because of the extra consumer surplus that non-market valuation studies seek to detect. Nevertheless, Pearce (2007) maintains that the divergence is too large to be explained by such arguments and that it is more likely to be attributed to censoring.

⁵ This includes some 60 individual benefit estimation studies and 40 cost studies, as well as over 130 case studies described in six survey reports by Chomitz and Kumari (1998), IIED (2003), Kramer *et al* (2003), Krieger (2001), Pearce and Pearce (2001), and Turner *et al* (2003). A database of case studies is available on request from the authors.

Where studies consider the costs and benefits of changes in biodiversity, these are prioritised. However, the majority of available case studies do not estimate the value of forest biodiversity directly but rather estimate the benefits or costs of protecting forests as a whole. These broader forest values are also considered here, where they relate to forests characterized by relatively high levels of biodiversity or forests that are unique in some other way, on the basis that the conservation of forest biodiversity and forest ecosystems more generally are complementary activities. We do not consider studies on the economics of planting and managing forests specifically for timber production, fuelwood, or for indirect benefits such as carbon sequestration or shelterbelts. Overall, our aim is to isolate those values which relate specifically to the diversity of forest ecosystems.

To the extent possible, we compare the benefits and costs of conserving forest biodiversity in different types of forest ecosystem, for example tropical versus temperate forests or old-growth versus newly-planted forest. We also examine how values vary across locations, which may have different levels of development or different cultural or institutional contexts. These variables frequently overlap, due to the geographical distribution of forest types. In addition, we focus on the *marginal* costs and benefits of forest biodiversity conservation, where estimates are available, and further distinguish between studies that estimate the values associated with increases in forest area or diversity, and those associated with deforestation or loss of biodiversity.

As well as reviewing the results of the case studies, this chapter also considers the extent to which values can be extrapolated or transferred from one particular study site to other contexts. The validity of such benefits transfer will depend on the physical characteristics of the forest ecosystem; the income, demographics, and preferences of the beneficiaries in a particular study; and the geographic scale over which values are applicable. Note that all cost and benefit estimates reported here have been standardized and are expressed in terms of US\$ values for the year 2000.

2.2 Case Study Evidence on the Benefits of Forest Biodiversity Conservation

2.2.1 Introduction

The case study evidence on the benefits of forest biodiversity conservation is reviewed here according to the type of economic value estimated. As set out in the preceding chapter, forest values may be categorised according to how they contribute to total economic value: i.e. direct uses, indirect uses, and non-use values.

Timber is not included here as a direct use value of forest biodiversity. This is because, while it is possible to harvest timber without destroying forest biodiversity, there is more often a trade-off between timber extraction and biodiversity conservation. Although the value of timber that can be harvested is part of the total economic value of an intact forest, in many cases it cannot be realised without sacrificing other biodiversity values. In such situations, timber values may be considered part of the opportunity costs of conserving forest biodiversity, and are therefore included here under the costs of biodiversity

conservation (section 2.3). In some cases, the benefits of forest biodiversity and of timber extraction can be obtained simultaneously, particularly if forests are managed sustainably. Where this applies, timber benefits may be complementary to other benefits of forest biodiversity and therefore enhance the possibilities for conservation. However, Pearce *et al* (2003) examine the profitability of sustainable forest management for timber production, which aims to ensure long term timber harvests as well as conserving non-timber benefits. They conclude that sustainable forest management is rarely as profitable as conventional harvesting regimes. Thus while sustainable management may provide some positive returns from timber, switching from conventional to sustainable forest management will generally entail a financial cost, in the form of reduced profits. Note also that the net cost of sustainable forest management for timber production may not be as high as other biodiversity conservation options, due to the income obtained from logging, but the benefits may also not be as great as if all extractive activity is avoided.

2.2.2 Direct Use Values

Non-timber Forest Products

Forests provide a range of products other than timber, including food for human consumption and forage for livestock; fibre for clothing or household objects; fuel for space heating and for cooking; and medicinal or cultural products. These non-timber forest products (NTFPs) are often harvested on a small scale from the wild, for direct subsistence use, although they may also be extracted on a larger scale or cultivated for commercial purposes. The case studies listed below in Table 1 include NTFP values from the collection of fuelwood and charcoal; plant products such as fruit, latex, oils, rattan and medicines; and animal products such as bushmeat, fish, eggs and honey.

The majority of the case studies listed in Table 1 obtained values for NTFPs through surveys of households involved in their collection. These surveys ask about the quantities of one or more products harvested by the household over a given time period, for subsistence use and/or sale. Where products are marketed, a financial value can be obtained directly, although harvest costs may need to be estimated in order to derive net values. Where products are not marketed, or a local market does not exist, values may be imputed in some way, possibly based on a close substitute, for example other fuels that could be used in place of fuelwood. Again, a value must be estimated for the time spent on harvesting the products, although in many of the case studies harvesting costs are not accounted for and the values presented are estimates of gross income. Table 1 specifies in each case whether the estimated values of NTFPs are gross values or net of harvesting costs and any processing or transport costs.

It is possible to measure the value of a forest for NTFP production on the basis of the existing stock; or the potential flow of NTFPs if harvests were optimal; or the actual flow based on existing harvesting patterns (which may be sustainable or unsustainable). Most of the case studies reviewed for this report estimate the value of actual flows of NTFPs.

These range from below US\$10/ha/year up to US\$330/ha/year, with a mean value of approximately US\$40/ha/year.

Table 1 – The value of non-timber forest products (NTFPs)

Location	Value of NTFPs (US\$/ha/year)		Reference
	Gross	Net	
Guatemala (stock of goods)	787		Ammour et al (2000)
Sri Lanka (stock of goods)	622	377	Batagoda (1997)
Peru (stock of goods)	700	420	Peters et al (1998)
Sri Lanka (potential flow)		186	Batagoda (1997)
Brazil (potential flow)		20	Pinedo-Vasques et al (1992)
Ecuador (potential flow)		200	Myers (1988)
Philippines	Actual flow: 65 Potential flow: 173		Saastamoinen (1992)
Global	50		Godoy et al (1993)
India	19-55		Murthy et al (2005)
India	122.5		Mahapatra et al (2005)
India	65		Verma (2000)
India	117-144		Chopra (1993)
India		70	Appasamy (1993)
Lao PDR	6-8		Rosales et al (2005)
Cambodia		19	Bann (1997)
Nepal		33-115	Houghton and Mendelsohn (1996)
Sri Lanka		14	Batagoda (1997)
Sri Lanka		13	Gunatilake et al (1993)
Sri Lanka		2	Batagoda (1997)
Malaysia		8	Caldecott (1988)
Guatemala		30	Ammour et al (2000)
Venezuela		15	Melnik and Bell (1996)
Ecuador		77-180	Grimes et al (1994)
Belize		41-188	Balickc and Mendelsohn (1992)
Mexico		330	Adger et al (1995)
Mexico		116	Alcorn (1989)
Brazil	79		Anderson and Ioris (1992)
Brazil	97		Mori (1992)
Venezuela		1	Thorbjarnson (1991)
Peru	67		Smith et al (1997)
Peru		18-24	Padoch and de Jong (1989)
Senegal	0.7		Ba et al (2006)
Cameroun		6	Yaron (2001)
Madagascar	4		Kramer et al (1995)
Kenya	88		Emerton (1999)
Uganda	11		Howard (1995)
Zimbabwe	21		Bojo (1993)
Zaire		1-3	Wilkie (1989)
Cameroun		1	Ruitenbeek (1988)
USA	3		Philips and Silverman (2007)
USA	25 for one deer 13 for second deer		Livengood (1983)
Mediterranean countries	39		Croitoru (2007)
Turkey	5		Bann (1998)
Scandinavia		10-15	Turner et al (2003)

Studies valuing the benefits of NTFPs may overstate the average value of maintaining forest biodiversity, because research is more likely to be carried out in locations where NTFPs are important to nearby communities. Less accessible forests will tend to exhibit lower values, as local demand for products will be lower and harvesting costs will be higher. In addition, because these studies generally measure the benefits arising from

forests as they are used at present, they do not provide information on whether current harvests are sustainable, or whether harvests could be increased with alternative management practices.

The vast majority of the studies carried out on the value of NTFPs focus on tropical forests in developing countries. It would be misleading to transfer these values to other forest types or to developed country contexts. A small number of studies examine the value of non-tropical forests for NTFP collection. For example, Bann (1998) estimates the gross value of NTFPs in Turkish forests at US\$5/ha/year, and Croitoru (2007) estimates the average value of NTFPs across all Mediterranean countries at US\$39/ha/year. Turner *et al* (2003) estimate the value of NTFP collection in Nordic forests at US\$10-15/ha/year, net of collection costs. These results suggest that NTFP values in temperate forests in developed countries are generally lower than in tropical regions, partly because fewer people rely on forest land for subsistence in the developed world. However, the relative dearth of studies does not mean that NTFP values do not exist for some temperate forests. For example, where mushroom collection, hunting or truffle harvesting are significant, NTFP values may be higher than suggested by the studies reviewed here.

Tourism/Recreation

Case studies valuing the recreational benefits of forests typically fall into one of two categories: i) temperate forests in developed countries used mostly by local residents for recreation, and ii) tropical forests in developing countries visited by foreign and sometimes also by domestic tourists.

Temperate forests/developed countries:

Several case studies estimate the value of forests in developed countries for recreational activities such as walking, fishing, hunting or wildlife viewing, mainly by local residents (Table 2). These studies focus on the USA and the UK, as well as some other European countries, and most look at temperate forests. Estimated values per trip are fairly low, at less than US\$5 in most studies. However, the annual values cited by Van der Heide (2007), Clinch (1999), Kramer *et al* (2003) and Gurluk (2006) are somewhat higher, ranging from US\$10-62 per trip. This would be consistent with local residents making multiple trips to nearby forest areas, but may also indicate that existence or option values are being included as well as direct use values.

Table 2 – Recreational values in temperate forests

Location	Value (US\$/ha/year)	Value (\$/trip or \$/household)	Reference
Ireland		All values WTP/hh relative to commercially managed Sitka spruce forest (median/mean) Pine forest: \$57 / \$31 Mixed forest: \$ 151 / \$42 Natural forest: \$162 / \$51	Mill <i>et al</i> (2007)
Ireland		\$1.4-3.6 per visit. Values \$0.2-0.6 higher if national park status conferred.	Scarpa <i>et al</i> (2000)
Scandinavia	\$15-20		Turner <i>et al</i> (2003)

Location	Value (US\$/ha/year)	Value (\$/trip or \$/household)	Reference
Denmark		Mean WTP/hh/year (Value relative to monoculture forest): Mixed forest: \$147 Deciduous: \$117 Selective felling: \$130 Screening: \$31 Some old trees: \$16	Olsen and Lundhede (2005)
Denmark		Value of 5-15% increase in Beech and Oak: \$29-154/hh	Aakerlund (2000)
Netherlands		\$52 per household (CV); \$0.05-0.39 per trip (TC)	van der Heide et al (2005)
Ireland	\$250	\$16/household	Clinch (1999)
Germany		\$42/person/year to visit all forests (day users). \$13/person to visit one forest during stay in region.	Ellasser (1999)
England		\$1.0 per visit	Bateman and Langford (1997)
England		\$0.5-2.0 per visit	Willis et al (1998)
England	\$2290		Bateman et al (1996)
Scotland		\$2.5 per visit	Hanley (1989)
UK		\$0.8-2.4 per visit	Willis and Benson (1989)
England		\$0.8-2.6 per visit	Bishop (1992)
Scotland, all forests		\$1.8-3.0 per visit	Hanley and Ruffell (1991)
UK, all forests		\$1.5-1.7 per visit	Hanley and Ruffell (1992)
UK		\$1.3-1.8 per visit	Whiteman and Sinclair (1994)
Italy	\$77-85		Bellu and Cistulli (1997)
Turkey		\$62/person/year	Gurluk (2006)
Turkey	\$0.4		Bann (1998)
Lebanon		\$38/household/year	Sattout et al (2007)
USA, national forests		Contribution to GDP: \$1244	Moskowitz and Talberth (1998)
USA, roadless areas of national forests		Economic impact: \$63 User day values: \$25	Loomis and Richardson (2000)
USA, Wisconsin	\$20-50		Scarpa et al (2000)
USA, Southern Appalachians	\$7582 (PC); \$26,498 (DC)	\$10/person	Kramer et al (2003)
USA, Washington		Mean WTP to suspend logging activities: \$87/hh	Cedar River Group (2002)
USA, Tennessee		Mean WTP to visit forest: \$170-242/person/year	Russell et al (2001)
USA, Montana	Elk hunting: \$3.5; Fishing: \$0.6	Elk hunting: \$108/trip,	Loomis (1992)
USA, Colorado and Virginia		Colorado: \$14/visitor day. Virginia: \$12/visitor day.	Walsh and Loomis (1989)
USA, Montana	\$110 direct spending		Yuan and Christensen (1992)
USA, Southern Appalachians	Hunting: \$6,500; Fishing: \$930-2,500		Moskowitz and Talberth (1998)
USA, Southern Appalachians	Fishing: \$1600; Hunting: \$2276; Wildlife viewing: \$1600; Recreation: \$23,000.		Barnhill (1999)
USA, Vermont		Median WTP: \$9.04-10.42/hh/year Total use value: \$29 million.	Gilbert et al (1992)
USA, Pacific Northwest	Hunting: \$14; Fishing: \$9.4		Moskowitz and Talberth (1998)

Location	Value (US\$/ha/year)	Value (\$/trip or \$/household)	Reference
Canada		\$52/person/year	McDaniels and Roessler (1998)

Recreational values per hectare of forest are extremely variable, ranging from less than \$1 to many thousands of dollars. This is partly due to the measurement of different things, for example, Scarpa *et al* (2000) estimate recreational values using the difference between actual and potential timber yields, on the assumption that forest owners would maximise their timber returns unless they obtain non-market benefits from the amenity or recreational use of the forest. The values they obtain are relatively low (US\$20-50/ha/year), but capture only the benefits to those making decisions about forest management and not to other potential forest users.

Two key determinants of recreational value per hectare across all of the studies are the accessibility of forests and the size of the local population. Loomis and Richardson (2000) estimate the value of recreational activities in roadless (i.e. difficult to access) areas of US National Forests at US\$25/ha/year, based on user-day values, or US\$63/ha/year, based on economic impact. These values are notably lower than estimates for recreation in National Forests more generally. Kramer *et al* (2003), Moskowitz and Talberth (1998), and Barnhill (1999) use different methods to estimate the value of recreation in the Southern Appalachian region of the USA, which is within one day's drive for about 120 million people (Kramer *et al*, 2003). They obtain very high values: between US\$930-2,500/ha/year for fishing alone, and over US\$20,000/ha/year for all forms of recreation.

The case studies listed in Table 2 mainly estimate the values of the simple availability of forest land for recreational use. Alternatively, some studies, such as Kramer *et al* (2003) and Sattout *et al* (2007) estimate the recreational value of maintaining a minimum level of forest quality. Other studies compare the values of more or less diverse forests for recreation by local users. Hanley *et al* (1998) find that UK households are willing to pay US\$22/year for a move from forests containing only evergreen trees to forests containing a mix of evergreen, larch and broadleaved trees, while Mill *et al* (2007) find that Irish households are willing to pay US\$51 more for recreation in natural forests relative to commercially managed Sitka spruce forests. In Denmark, Olsen and Lundhede (2005) find that households are willing to pay between US\$16 and US\$147 more for varying increases in diversity relative to a baseline of monoculture forest. An exception to the general pattern of positive preferences for greater forest diversity is provided by Horne *et al* (2005), who report that, while people in Finland express positive non-use values for forest biodiversity, they prefer more managed forest areas, with lower levels of species richness, for recreational purposes.

Tropical forests/developing countries:

A second category of case studies includes those that estimate the values of tourist visits from abroad, or from within the country, to tropical forests located in developing countries. These values may express the consumer surplus enjoyed by tourists, or they may reflect the capture of consumer surplus, for example through park entrance fees or increased economic activity, and the resulting benefits to government or local communities.

The estimated values of tourism in tropical forests are around US\$10-50 per visit, significantly higher than the corresponding values for temperate forests (Table 3). This is most likely due to differences in the nature of forest recreation: values for temperate forests reflect the benefits of short visits by local residents to nearby forests, while values for tropical forests reflect the benefits of longer visits to more distant destinations. Nevertheless, the magnitude of the estimated values, as well as the fact that people are clearly willing to travel long distances to visit tropical forests, indicate that they gain significant benefits from the specific characteristics of such forests, which often include high levels of biodiversity. Ideally, travel cost estimates of tourism benefits from tropical forests would include the cost of reaching the country as well as travel within the country. However, joint consumption of forest-related benefits and other benefits of visiting a tropical country, such as warm weather, culture or beaches, mean that the proportion of total travel costs that can be attributed to forests cannot be easily identified.

In studies such as Bienabe and Hearne (2006), Van Beukering *et al* (2003) and Schultz *et al* (1998), which estimate the values per visit for both foreign and domestic tourists, the authors find that recreational values are consistently higher for foreign tourists than for local visitors. This is hardly surprising, given the higher average incomes of foreign tourists visiting developing countries.

Table 3 – Recreational values in tropical forests

Location	Value (US\$/ha/year)	Value (\$/trip or \$/household)	Reference
Uganda (foreign tourists)	\$0.59 with 20 bird species; \$1.32 with 80 bird species	\$46/person	Naidoo and Adamowicz (2005)
Madagascar (foreign tourists)	\$10.73 (TC); \$29 (CV - may include existence value)	\$27/trip (TC); \$74/trip (CV - may include existence value)	Kramer <i>et al</i> (1995)
Madagascar (foreign tourists)	\$360-468		Maille and Mendelsohn (1991)
Indonesia (foreign and local tourists)		\$7.11 per visitor (local tourists); \$12.4 per visitor (foreign tourists)	van Beukering <i>et al</i> 2003
Malaysia	\$3		Bann (1999)
Malaysia (foreign tourists)	\$740		Garrod and Willis (1997)
Thailand (foreign tourists)	Tourism expenditure: \$7 35.5/ha/year; consumer surplus: \$2.3/ha/year		Dixon and Sherman (1990)
India		WTP \$2.76/household/year	Hadker <i>et al</i> (1997)
Costa Rica (foreign and local tourists)		WTP for '1 level' increase in scenic beauty: Costa Ricans - \$2.93/year ; Foreign tourists - \$3.28	Bienabe and Hearne (2006)
Costa Rica, Two forested parks (foreign and local tourists)	\$950 and \$2305 (two sites).	\$11 and \$13 per local visitor, and \$23 and \$14 per foreign visitor.	Shultz, Pinazzo and Cifuentes (1998)
Costa Rica, 3 national parks (foreign tourists)		\$21-25 per visitor	Chase <i>et al</i> (1998)
Costa Rica (foreign tourists)	\$160		Tobias and Mendelsohn (1991)
Costa Rica (foreign and local tourists)		\$60/visit (current fees \$30/visit)	Baldares <i>et al</i> (1990)
Bolivia (foreign tourists)	\$2.4-2.8/ha/year	Mean WTP: \$72 (CB); \$35 (CV)	Ellingson and Seidl (2007)

Mexico	\$1	Adger et al (1995)
Brazil, Atlantic Coastal Forest	WTP for new parks: \$23-\$89/person WTP for protection of half of remaining forest: \$9/person	Holmes et al (1998)

As with temperate forests, the estimated value of recreation varies enormously across studies of tropical forests, from less than US\$1 to over US\$2000 per ha/year. Higher values are observed for sites of special scenic interest, such as the Costa Rican forest parks studied by Schultz *et al* (1998), and for more accessible areas, such as the Malaysian site studied by Garrod and Willis (1997).

Per hectare values of recreation also vary depending on the political situation in the country or region in which a forest is located. Van Beukering *et al* (2006) estimate the value of consumer surplus for tourists visiting the Leuser Ecosystem in Northern Sumatra at US\$7-12 per person. In 1999, about 8,000 tourists visited the area, which implies a total annual value from US\$56,000-96,000. However, the authors note that the number of tourist visits when the study was carried out had declined relative to earlier years, due to the deteriorating regional security situation. If tourist numbers had remained at the 1995 level of 25,000 visits, the total recreational value of the Leuser Ecosystem would have been in the range of US\$175,000-300,000.

Bioprospecting

A potentially significant value of forest biodiversity is as a source of genetic information for the development of new agricultural crop varieties, new medicines, or other industrial products and processes. Investor interest in realizing such ‘bioprospecting’ values, particularly in relation to the pharmaceutical industry, is demonstrated by some recent agreements between private companies and countries harbouring diverse tropical forests.

One of the most famous examples is an agreement signed in 1991 between INBio, a private, non-profit, scientific organization established by the Costa Rican government, and Merck, a US multinational pharmaceutical corporation. In return for an upfront payment, training assistance and a promise of royalties on future sales of products derived from Costa Rica’s forests, INBio agreed to supply Merck with samples of plants, insects and micro-organisms collected from the wild. Merck thus secured the right to use these samples to develop new pharmaceutical products. This example has stimulated interest in the possibility of developing new markets for the genetic information provided by forest biodiversity. Optimism about the potential of such markets has been dampened more recently, as additional bioprospecting agreements have been slow to emerge. This may be partly due to the slow progress of diplomatic efforts to agree an international framework for securing access to genetic resources and sharing the benefits thereof.

Table 4 – Bioprospecting values

Location	Value (US\$/ha/year)	Reference
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Biodiversity hotspots	Random search, locations with highest biodiversity: Value for bioprospecting: \$1.09 - \$265/ha depending on parameters used in model. Ordered search, most promising locations: Value for bioprospecting: \$12-\$58/ha	Costello and Ward (2006)
Biodiversity hotspots	Range from \$0.2 per hectare in California Floristic Province to \$20.6 per hectare in Western Ecuador.	Simpson et al (1996)
Biodiversity hotspots	Range from \$29 per hectare in California Floristic Province to \$2888 per hectare in Western Ecuador.	Craft and Simpson (1996)
Biodiversity hotspots	Range from \$0 per hectare in California Floristic Province to \$9177 per hectare in Western Ecuador.	Rausser and Small (1998)
Lao PDR	\$0.1-0.52/ha/year	Rosales et al (2005)
Mexico	\$6.4/ha/year	Adger et al (1995)
Malaysia	\$0.52-695/ha/year	Kumari (1995)

The value of forest genetic information is difficult to quantify in relation to marginal changes in forest area, because of uncertainty regarding the extent to which such information is distributed spatially. Much of the genetic material that occurs in one location may also be present elsewhere, suggesting that until stocks of forest biodiversity are severely reduced, marginal values will remain low. In addition, although returns are potentially very high, if a new product is developed, the probability of finding commercially valuable material from any one biotic sample remains extremely low.

Early studies, such as those by Adger *et al* (1995) and Kumari (1995), estimated the values of bioprospecting by multiplying the probability that a commercially-valuable substance would be found by the value of the product to the pharmaceutical company or to government. However, both the probabilities and the resulting sales or royalty revenues are based on many assumptions that are difficult to verify. Furthermore, this approach provides average values of bioprospecting for a particular forest, rather than the marginal value of avoiding deforestation.

More recent studies estimate the marginal value of a species (or of the land providing habitat for a species) using data on search costs, the probabilities of success and potential revenues from successful products. As these methods also involve many assumptions about the relevant parameters and, crucially, about the search methods used by pharmaceutical companies, the results vary considerably for the same forest locations. One such study, by Simpson *et al* (1996), found that even in so-called biodiversity hotspots, marginal values were relatively modest, at around US\$20/ha/year. In contrast, Rausser and Small (1998) estimated bioprospecting values ranging from US\$0 to over US\$9,000 per hectare. Costello and Ward (2006) investigated the reasons why these studies arrived at such different results and conclude that it is mainly due to differences in the assumptions about key parameter values. They re-estimate the same models using alternative parameter values (which they consider defensible), and report marginal values ranging from US\$1-265/ha/year.

The values reported above are all for biodiversity hotspots – the vast majority of forested areas may have relatively little value for bioprospecting. Note also that most of these studies estimate the private returns to bioprospecting, and thus implicitly account for the fact that profits from new drugs tend to decline over time, as they are superseded or as

patents expire. Simpson and Craft's (1996) results, on the other hand, suggest that social values due to improved healthcare will be considerably higher than private values.

2.2.3 Indirect Use Values

The indirect use values of forest biodiversity are based on the existence of forest ecosystems in good ecological condition. Crucially, however, those who enjoy the benefits of indirect use values are not necessarily aware of this dependence. Examples of the indirect use values of forest biodiversity include watershed protection services, such as flood prevention and water purification, carbon sequestration and assimilation of other pollutants, and pollination of agricultural crops and other plants. The case study estimates of indirect use values are presented in Table 5 and discussed further below.

Watershed protection

Watershed protection services depend primarily on the presence of trees or other vegetation and are not necessarily related to the diversity of forests. However, there may be an indirect relationship between forest biodiversity and watershed protection. First, as discussed earlier, the conservation of biodiversity provides insurance against the risk of ecosystem collapse. Second, watershed protection benefits may strengthen the incentives to conserve intact forests, rather than harvesting timber or converting land to other, potentially less diverse uses. For these reasons, we examine the watershed protection benefits of conserving natural forests.

The loss or degradation of forest cover can have detrimental impacts on watershed functions. These include changes in water flow regulation, which can result in flooding or storm damage; and increased soil erosion, with resulting siltation and sedimentation of rivers, reservoirs and other water bodies, as well as loss of nutrients in soil used for agriculture. Such impacts have economic consequences, although they may not always be significant. Economic effects include damage to agricultural land or residential property, due to flooding; reductions in the productivity of agricultural land; increased water treatment costs or loss of storage capacity in reservoirs; and damage to equipment used in hydroelectric facilities.

Table 5 – Indirect use values of forests

What is being valued	Location	Value (US\$/ha/year)	Reference
Cost of soil replacement and preventing soil loss.	Guatemala	Negligible for soil loss; \$12/ha for nutrient loss; \$30/ha for NTFPs and ecosystem services.	Ammour et al (2000)
Sedimentation effects on infrastructure	Mexico	Negligible	Adger et al (1995)
Watershed protection functions	Lao PDR	\$309-1576/ha/year	Rosales et al (2005)
All ecosystem services	India	\$4348/ha/year	Verma (2000)
Shoreline protection and fisheries protection by mangrove forest.	Malaysia	Shoreline protection: \$845/ha. Fisheries protection: \$526/ha.	Bann (1999)
Impacts of RIL on hydroelectricity	Malaysia	\$4/ha	Shahwahid et al (1997)
Protection of irrigation	Malaysia	\$15/ha for irrigation water.	Kumari (1996)
Indirect uses of forests	Malaysia	\$20-23m/year	Bennett and Reynolds (1993)
Drought mitigation	Indonesia	WTP: \$4.10/ha/year; Estimated profit foregone: \$6.80 /ha/year	Pattanayak and Kramer (2001)
Watershed protection benefits	Philippines	\$223-455/ha/year	Paris and Ruzicka (1991)
Fisheries protection	Philippines	\$268/ha	Hodgson and Dixon (1988)
Flood protection	Cameroun	\$0-24/ha	Yaron (2001)
Flood protection	Cameroun	\$3/ha	Ruitenbeek (1989)
Watershed protection	Kenya	\$273/ha/year	Emerton (1999)
Watershed protection	Uganda	\$4.63/ha/year	Howard (1995)
Replacement costs of soil nutrients	Turkey	\$46/ha	Bann (1998)
Watershed protection functions - water supply. Carbon sequestration.	Ireland	Watershed protection: -\$20/ha (negative amount). Carbon sequestration: \$88 per ha.	Clinch (1999)
Watershed protection functions	USA, Hawaii	\$1022/ha/year	Kaiser and Roumasset (2002)
Consumptive use of all water flowing from forests	USA	\$90/ha/year	Dunkiel and Sugarman (1998)
Indirect uses of forests	Canada	\$64/person/year	McDaniels and Roessler (1998)
Shelterbelts for crop protection and farm forestry	Northern Nigeria	Rate of return increases from 5% to 13-17%.	Anderson (1987)
Bee pollination for coffee production	Costa Rica	\$361/ha/year	Ricketts et al (2004)
Gain in profits to rice and coffee production	Eastern Indonesia	\$3-35 per household	Pattanayak and Kramer (2001)
Pollution removal by trees in urban areas	USA	\$447-663/ha/year	Nowak et al (2007)
Carbon sequestration	USA	\$58.8/ha/year	Loomis and Richardson (2000)
Carbon sequestration	Canada	\$24-120/ha/year	Van Kooten and Bulte (1999)
Carbon sequestration by US national forests	USA	\$37/ha/year	Dunkiel and Sugarman (1998)
Carbon sequestration	UK	\$280-413 per ha	Pearce (1994)
Carbon sequestration	Scandinavia	\$10-15/ha/year	Turner et al (2003)
Carbon sequestration	Uganda	\$5.83/ha/year based on damage costs; \$6.81/ha/year based on replacement costs	Howard (1995)
Carbon sequestration	Costa Rica	\$105/ha/year	Bulte et al (2002)

Case studies that estimate the value of forests for watershed protection typically rely on production function approaches, in which the downstream impacts of changes in forest quality or extent are assigned a financial value. This generally involves making certain

assumptions about the relationship between forest condition and downstream activities, which is not always well understood.

The values reported in the case studies reviewed here suggest that, in certain cases and particularly in tropical forest areas, watershed protection values can be high. Rosales *et al* (2005), Bann (1999), Paris and Ruzicka (1991), Emerton (1999) and Kaiser and Roumasset (2002) estimate values ranging between US\$200/ha/year and about US\$1,000/ha/year. Most of these studies present combined values for a range of watershed functions, for example soil conservation, reduced flooding risk, maintenance of fisheries, and avoided damage to hydropower facilities. Other studies find lower values, ranging between US\$0-50/ha/year, although these mainly consider individual watershed functions such as avoiding soil erosion (Ammour *et al*, 2000), regulating water supplies (Kumari, 1996) or avoiding flood damages (Yaron, 2001; Ruitenbeek, 1988). Finally, there are some cases in which increased forest cover appears to reduce downstream benefits; Clinch (1999) values watershed protection functions at **negative** US\$20/ha/year, on the grounds that forest cover reduces the volume of water flowing downstream.

Estimated watershed protection values cannot easily be transferred across forest areas, because they depend on site-specific human uses, soil and water conditions, and the climate of the particular watershed. In general, benefit transfer can only be reliably undertaken for sites having very similar characteristics.

Carbon sequestration

There is growing interest in the value of forests for carbon sequestration, particularly as deforestation is understood to be a significant contributor to global emissions of greenhouse gases (Stern, 2006). As with watershed protection, carbon sequestration benefits do not necessarily depend on the diversity of forest ecosystems. Nevertheless, different types of forest have the potential to store different amounts of carbon. Houghton (1999) reports that undisturbed, temperate evergreen and deciduous forests store an average of 160t and 135t of carbon per hectare, respectively, while moist tropical forests can store 250t per hectare or more. In contrast, grassland stores 7-20t of carbon per hectare, on average. As a result, in addition to providing incentives to conserve forests in general, the variation in carbon storage capacity tends to reinforce most especially the incentive to conserve tropical forests.

Case studies of the benefits of carbon storage in forests typically compare the amounts of carbon stored under alternative land use scenarios and then place a monetary value on the difference. Such estimates are highly sensitive to the assumptions made about the market price or damage costs associated with a tonne of carbon emissions, as well as the carbon sequestration process and (because sequestration and climate change are slow processes) the discount rate. Accordingly, the values obtained vary from less than US\$10/ha/year to over US\$400/ha/year. Van Kooten and Bulte (1999) estimate the value of carbon storage in Canadian forests at between US\$24-120/ha/year. Bulte *et al* (2002) estimate an average value of \$102/ha/year for Costa Rican forests, using a discount rate of 7% and assuming a carbon price of \$10/tonne.

Other indirect uses

Forests provide many other valuable ecological services, besides watershed protection and carbon sequestration. For example, Ricketts *et al* (2004) examine role of wild pollinators and find that the value of forest patches as habitat for bees is US\$361/ha/year, in terms of improved yields on neighbouring coffee farms (within 1km). Pattanayak and Kramer (2001) consider the benefits of forest biodiversity for coffee production, as well as rice production, and estimate these at US\$3-35 per farming household. Nowak *et al* (2007) estimate the value of air pollution removal by trees in urban areas at US\$447-663/ha/year.

2.2.4 Non-Use Values

Non-use values include both existence and bequest values. In practice, it is extremely difficult to separate these two categories of value. The studies reviewed below all estimate total non-use values, with only a few attempting to distinguish between individual component values. The estimates were all obtained using contingent valuation methods; it is not always certain that only non-use values are elicited, as respondents may also consider use values such as recreation in their responses. This effect is reported by Garrod and Willis (1997), for example, despite the fact that they specifically asked respondents to value forests that were remote and rarely visited.

Table 6 – Non-use values of temperate forests

What is being valued	Location	Value (US\$/ha/year)	Value (US\$/household/year)	Reference
Non-use values of biodiversity	UK	\$891-2427	WTP per household for increase of 12,000ha: Upland conifer: \$.49; Lowland conifer: \$.41; Lowland ancient semi-natural broadleaved: \$.159; Lowland new broadleaved: \$.118; Upland native broadleaved: \$.127; Upland new native broadleaved: \$.86.	Hanley <i>et al</i> (2002)
Increased biodiversity protection in remote forest areas	UK	Low diversity: \$3899-4299 Medium diversity: \$6653-7258 High diversity: \$2381-2663	WTP for 3000ha increase Low diversity: \$.0506-0.558 Medium diversity: \$.0864-0.943 High diversity: \$.0309-0.346	Garrod and Willis (1997)
Preserve endangered species	Norway		All endangered species in Norwegian forests: \$.91-150	Veisten <i>et al</i> (2003)
Natura 2000 programme	Finland		3% increase in area: \$158/hh 3% decrease in area: \$692/hh	Li <i>et al</i> (2004)
Increased conservation of forest land	Finland		Mean WTP for increase in conservation area from 1.8% to 4.2%: \$15/household	Horne <i>et al</i> (2004)
Forest conservation	Finland		Mean WTP: \$214/hh (CV); \$119-214/hh (CE)	Lehtonen <i>et al</i> (2003)

			Marginal WTP for reduction in endangered species: -\$0.302/hh Reduction in endangered species from 650 to 300: \$119/hh	
Protection of old-growth forest	Finland	WTP: \$337	Mean WTP: \$50/person/year	Knivila et al (2002)
Increased conservation in biodiversity hotspots	Finland	Proportion of hotspots: 25%: \$670-700 50%: \$390-420 75%: \$110-190	Proportion of hotspots: 25%: \$45-47 50%: \$73-78 75%: \$33-56	Siikamaki and Layton (2007)
Preserving tea tree woodlands	Australia		Mean WTP for preserving teatree woodlands: \$12/ha/year	Mallawaarachi et al (2001)
Protection of Carmel National Park	Israel	Total existence value estimated as \$ 2,324/ha/year	CV results: Mean WTP to avoid fire damage - \$42.3 Donor results: Mean WTP to avoid fire damage - \$60.8	Schechter et al (1998)
Existence values of north forests of Iran	Iran		Mean WTP to reduce deforestation: \$28/year	Amirnejad et al (2005)
Non-use values of wild forest land	USA, Alaska	\$24/ha/year		Phillips and Silverman (2006)
Changes in biodiversity conservation	USA, Oregon		Old growth forest: \$388 Endangered species habitat: \$255 Salmon habitat: \$147 Biodiversity reserves: \$46	Garber-Yonts et al (2004)
Protection of high elevation spruce fir forest	USA, Southern Appalachians	Median WTP: \$33,243/ha/year (PC), \$116,182/ha/year (DC)	Median WTP for all remaining forest: \$19	Kramer et al (2003)
Restoring old-growth longleaf pine forests	USA, South Carolina		OE: \$11 per year; PC: \$8 per year; DC: \$13 per year	Reeves et al (1999)
Habitat of the Mexican spotted owl	USA	\$4400/ha	\$102 per US household per year.	Loomis and Ekstrand (1998)
Avoided fire risk in California and Oregon forests	USA, California and Oregon	\$1.9-9.9 million/ha for all US residents	\$56 per household in California and New England.	Loomis and Gonzales-Caban (1997)
Reducing fire hazard to old growth forests	USA		OE: \$33 per year; DC: \$98 per year	Loomis et al (1996)
Forest quality (avoided infestation and air pollution)	USA, Southern Appalachians		\$82 per household	Haefele et al (1992)
Protection of mixed-age American larch pine	USA, Colorado		\$34/year	Walsh et al (1990)
Existence value of wilderness in Colorado	USA, Colorado	\$12-45/ha (lowest is for last increments)	Option value: \$16 per household; Existence+bequest value: \$38 per household	Walsh et al (1984)
Non-use values of forests	Canada		\$40/person/year	McDaniels and Roessler (1998)

Most of the case studies that elicit non-use values focus on temperate forests in developed countries, primarily the USA, the UK and Scandinavia. The values reported are generally higher than other values obtained for forest land, at over US\$1000/ha in several cases. Part of the reason for the relatively high estimates of non-use values is that WTP or WTA is often extrapolated over a large population, reflecting the public good nature of non-use benefits. Among studies that assess relatively large areas of forest considered significant in some way to the domestic population, WTP to preserve the forest averages around

US\$50 per household. The highest values reported are for forests that provide habitat for charismatic species, such as the Mexican Spotted Owl (Loomis and Ekstrand, 1998).

Table 7 – Non-uses values of tropical forests

What is being valued	Location	Value (US\$/ha/year)	Value (US\$/household/year)	Reference
Increased forest cover and biodiversity	China		Increase in forest and grassland cover: Beijing - \$97/year; Xi'an - \$38/year; Ansai - \$34/year Biodiversity: Beijing - \$0.36 per additional species protected; Xi'an - \$0.10; Ansai - \$0.04.	Wang et al (2007)
Protection of Guizhou snub-nosed monkey.	China		Mean WTP: \$1.27 (rural); \$3.82 (urban)	Gong (2004)
Maintain habitat of Giant Panda (value to OECD citizens)	China	value of maintaining Wolong Reserve: \$259/ha/year	Panda conservation in natural habitat: \$15.40 Difference in WTP for natural habitat rather than pens: \$6.67	Kontoleon and Swanson (2003)
Preserving forest in Korean de-militarised zone.	Korea		Mean WTP: \$15.42/person	Lee and Mjelde (2007)
Value of biodiversity conservation to local households	India		WTP for biodiversity conservation: \$130.5/household/year	Ninan and Sathyapalan (2005)
WTP of Sri Lankans for the forest reserve	Sri Lanka		Use values: 0.5% of income for peripheral villages, 0.2% for rural residents and 0.3% for urban residents. Bequest values: 0.4%, 0.1% and 0.2%. Existence values: 0.2%, 0.3% and 0.2%.	Gunawardena et al (1999)
Value of Khao Yai national park	Thailand		WTP for existence of elephants: \$7/person	Dixon and Sherman (1990)
WTP of Australian tourists for rainforest in Vanuatu.	Australia	\$53/ha/year	\$15/person	Flatley and Bennett (1996)
Preserve biodiversity	South Africa	\$21.76/ha/year		Turpie (2003)
Existence value of tropical rainforests for US citizens	Global	\$4.6/ha/year (Pearce and Pearce, 2000)	Payment card: \$31 per year; Dichotomous choice: \$21 per year.	Kramer and Mercer (1997)
Increased biodiversity protection	Costa Rica		WTP for '1 level' increase in biodiversity protection: Costa Rican residents - \$3.87; Foreign tourists - \$6.62	Bienabe and Hearne (2006)
Protection of the Brazilian Amazon (WTP of UK and Italian citizens)	Brazil	Mean WTP for protection of 5% more of the Brazilian Amazon: \$43/ha/year	Mean WTP for protection of 5% more of the Brazilian Amazon in the UK and Italy: \$42/ha/year	Horton et al (2003)
Existence value of Mexican forests	Mexico	\$0.03-10/ha/year		Adger et al (1995)

In the case of tropical forests, two different sets of values are of interest: those expressed by local populations and by foreigners. Flatley and Bennett (1996), Kramer and Mercer (1997), Horton *et al* (2003) and Bienabe and Hearne (2006) estimate the non-use values of tropical forests expressed by people resident elsewhere in the world. They find that

people living in countries *without* tropical forests nevertheless value the continued existence of these forests, with stated WTP ranging from around US\$7-42 per person/year or per household/year for the conservation of biodiversity, or of tropical forests more generally. Dixon and Sherman (1990) and Kontoleon and Swanson (2003) estimate the WTP of foreigners for protecting particular charismatic species (elephants and pandas respectively) and report broadly similar values to those for tropical forests as a whole. The Kontoleon and Swanson study further distinguishes the value to OECD citizens of preserving panda bears in pens from the value of conserving the species in its natural habitat, estimating the difference at approximately US\$7/person/year.

A few studies examine the non-use values expressed by local residents for tropical forests in their own country. Bienabe and Hearne (2006) find that residents of Costa Rica hold positive values for increased biodiversity in their nation's forests. Wang *et al* (2007) estimate local residents' values for increased forest and grassland cover in China at US\$56 per household per year, while Ninan and Sathyapalan (2005) find that Indian residents' average WTP for improved biodiversity conservation is US\$130.5 per household per year. These estimates are relatively high, compared with existence values expressed by people in developed countries, especially as a share of average income. However, they may incorporate some use values as well as existence values.

Most of the studies reviewed here examine the existence values of forests rather than of forest biodiversity. However, most case studies focus on forests that are either relatively unique to the region in question, or more diverse than the alternative land use options considered. For example, Mallawaarachi *et al* (2001) and Kramer *et al* (2003) analyze the existence values of tea tree woodlands and high elevation spruce fir forest, respectively, while Kniivila *et al* (2002), Reeves *et al* (1999) and Loomis *et al* (1996) estimate the values of old-growth forests. In studies of WTP for tropical forests in general (Kramer and Mercer, 1997), or for the Brazilian Amazon as a whole (Horton *et al*, 2003), it may be argued that a significant part of the value of these areas to foreign citizens reflects their diversity relative to other types of forest.

Part of the value of maintaining diverse forest land is as habitat for endangered species. Veisten *et al* (2003) estimate the average WTP to protect all endangered species in Norwegian forests at US\$91-150 per household/year, while Li *et al* (2004) find that Finnish households would experience a welfare increase of US\$119 per year from a reduction in the number of endangered forest species from 650 to 300.

It can be difficult to compare directly the values relating to particular forest types with the values for forests more generally, especially when they are derived from separate studies. However, several studies specifically ask respondents to compare more and less diverse forest types, or changes in forest biodiversity. Hanley *et al* (2002) elicit contingent values for six different forest types and find that households are willing to pay US\$1.59 for a 12,000ha increase in lowland ancient semi-natural broadleaved forest, compared to only US\$0.41 for the same increase in lowland conifer forest. Bienabe and Hearne (2006) find that Costa Rican and foreign residents are willing to pay US\$3.87 and \$6.62 per year, respectively, for an increase in biodiversity conservation, while in Finland,

Li *et al* (2004) find that households are willing to pay US\$158/year for a 3% increase in the area under Natura 2000, an EU-wide biodiversity conservation programme.

Garber-Yonts *et al* (2004) consider various options for increasing forest biodiversity, and find that households are willing to pay US\$255/year for a marginal increase in endangered species habitat and US\$388/year for an increase in old growth forest from 5% to 35%, although only US\$46/year for biodiversity reserves in which all economic activity is forbidden. The latter result may be because survey respondents sometimes account for the trade-off between increased biodiversity and lower timber returns in their valuations. Garrod and Willis (1997) find a 70% increase in WTP for conifer forest with some management for biodiversity, relative to conifer forest managed solely for timber. However, they find that WTP is lower for conversion to native woodland, which would offer no significant timber values. Similarly, Siikamaki and Layton (2007) find that Finnish households are willing to pay US\$73-78/year for an increase in the conservation of biodiversity hotspots from 10% to 50%, but their WTP declines to US\$33-56/year for an increase in hotspot coverage from 10% to 75% of total forest area.

2.2.5 Summary of the Benefits of Conserving Forest Biodiversity

The benefits of conserving forest biodiversity may be estimated in terms of various use and non-use values, which together comprise total economic value. The extent to which empirical estimates can be found for each type of value varies according to the difficulty of obtaining the value and the relative importance of the value for particular forest types or locations.

Estimates of direct use values are widely available, although these frequently relate to forests as a whole rather than the diversity of forests. The types of direct use value that are reported in the literature vary according to the use of particular forests. Thus we find significantly more information on the value of forests for NTFP collection in developing countries, where NTFPs are an important part of subsistence livelihoods, than in temperate forests in developed countries, where NTFP collection is often no more than an occasional, recreational activity. An exception to this is reported in some Mediterranean countries, where NTFPs are harvested commercially (Croitoru, 2007).

Across the large number of studies of NTFP collection in developing countries, values vary widely but largely fall within a range of less than US\$1 up to about US\$100 per hectare per year, averaging around US\$40/ha/year. None of the studies directly compare values from more or less diverse forests, although many of them list a wide range of forest products collected from the same forest locations. This suggests that the diversity of plants and animals occurring in a forest may be important to those collecting NTFPs.

Recreational uses of forests are valued at around US\$5 per trip in temperate regions and US\$10-50 per trip in tropical regions. The higher values for tropical forests, and the fact that many of those visiting tropical forests have travelled a long distance to do so, suggest that recreational benefits in tropical forests are greater than in less diverse temperate

forests. However, in many cases the benefits being valued are not directly comparable, because trips to tropical forests tend to last longer than trips to temperate forests. Per hectare values of forest recreation also vary widely, depending on how many visitors a particular forest receives. Key determinants of recreational value include the accessibility of the forest, the size of the local population, and whether the forest has unique features. The last of these factors will include high biodiversity or the presence of unique or endangered species, but also other features such as mountain views.

There has been much debate about the value of protecting forest biodiversity as source material for bioprospecting. Recent work on this question, by Costello and Ward (2006), suggests that, under defensible assumptions, the value of biodiversity hotspots for bioprospecting ranges between US\$0 and US\$265/ha/year. This indicates that a small number of highly diverse or unique forest sites will have values sufficiently large to justify protection as a source of genetic material for pharmaceutical research, while the majority of locations will have practically no bioprospecting value at present.

The main indirect use values of forested land reported in the literature relate to watershed protection and carbon sequestration. These values depend on the presence of forest as opposed to other forms of land cover and may not relate specifically to biodiversity, although carbon sequestration may be greater in some more diverse forest types. Values for combined watershed protection and climate regulation functions in individual case studies are generally high, ranging from US\$200/ha/year to over US\$1000/ha/year. However, watershed protection values are highly dependent on geophysical and climatic conditions and the size of the affected population, and can only be transferred to other locations with very similar characteristics. The value of forests for carbon sequestration likewise varies depending on the amount of carbon stored by different forest types, the value of each tonne of CO₂, and the discount rate. Fairly conservative estimates suggest that the climate benefits from afforestation or avoided deforestation are in the range of US\$100-200/ha/year.

Non-use values for temperate forests average around US\$50 per household per year, while non-use values held by residents of developed countries for tropical forests range from US\$7-42 per household per year. In both cases, values are higher for forests containing endangered species. A few studies which examine non-use values of tropical forests expressed by residents of the country in which these forests are located report relatively high values, although the estimates may partly reflect use values also. Studies that directly examine the impact of forest diversity on non-use values find significant relationships. For example, Hanley *et al* (2002) report that the non-use values expressed by UK residents for relatively diverse forests are up to four times greater than for low diversity conifer forest. Other studies find that, in some cases, households are willing to pay over US\$100 per year for increases in the diversity of forests or the protection of endangered species.

2.3 Case Study Evidence of the Costs of Forest Biodiversity Conservation

2.3.1 Opportunity costs of land

The largest component of the costs of forest biodiversity conservation is usually the cost of forgoing the alternative land uses. Estimates from case studies vary depending on the value of the alternative uses that are considered. For example, in Madagascar, several studies have estimated the costs to local households of being prevented from using the forest for activities such as NTFP collection (Caret and Loyet, 2003; Ferraro, 2002; Kramer *et al*, 1995). These costs range between US\$1.70/ha/year and US\$29/ha/year, as the majority of the activities that would take place in the absence of conservation are relatively low value. In contrast, the opportunity cost of not converting land to agriculture is estimated at US\$368/ha/year in some parts of Kenya (Emerton, 1999), while in certain areas of India, the opportunity cost of not converting land to coffee production is estimated at between US\$251-489/ha/year (Ninan and Sathyapalan, 2005).

In theory, the opportunity costs of biodiversity conservation should be expressed in terms of the highest value alternative use of the land. In many cases, the relevant land use is conversion to agriculture. For example, Naidoo and Ricketts (2006) estimate the average probability of agricultural conversion, multiplied by the benefits from conversion, at US\$2.70/ha/year across the Mbaracayu Biosphere Reserve in Paraguay. In other cases, the highest value alternative to conservation will be logging. Butry and Pattanayak (2001) estimate the costs to logging communities of the Ruteng National Park in Indonesia at US\$24/ha/year. Alternatively, a range of activities can be compared, using the highest value option for each individual parcel of land. Thus Chomitz *et al* (2005) estimate the opportunity costs of conservation for multiple parcels of land in the Amazon rainforest, in Bahia, Brazil. They report a median value of US\$16/ha/year, declining to an average of US\$6/ha/year for the 10,000 hectares considered least suitable for other uses.

Across the range of potential alternative uses for tropical forest land, opportunity costs will clearly vary, but are generally less than US\$100 per hectare per year, and in many cases below US\$5 per hectare per year. Grieg-Gran (2006), for example, reviews the opportunity costs of avoiding deforestation in eight tropical countries and reports costs ranging between US\$38/ha/year in Cameroon and US\$89/ha/year in Papua New Guinea.

Case studies from developed countries suggest significantly higher opportunity costs of conserving forest biodiversity, although this varies depending on local land scarcity and potential alternative uses. In European countries, high land values result in large estimates of opportunity costs. Siikamaki and Layton (2006) surveyed non-industrial private forest land owners in Finland to elicit their WTA for biodiversity improving management practices, and report median WTA of US\$738 per forest site. Similarly, in Denmark, Strange *et al* (2006) investigated the cost of species preservation on a range of land types and found that, to preserve 740 out of 763 priority species, the opportunity costs would fall between US\$412-638/ha/year.

In less densely populated developed countries, the opportunity costs of conservation are lower. Shaik *et al* (2007) found that farmers in parts of Canada were willing to accept US\$12/ha/year, on average, to convert to agroforestry, while Sinden (2004) estimated the opportunity costs to farmers in Australia of not converting native vegetation to farmland at US\$4-7/ha/year, depending on how much land they would be likely to convert in the absence of restrictions.

At a global level, Lewandrowski *et al* (1999) estimate the reduction in GDP that would result if 5% or 15% of land was retired from production and devoted to conservation⁶. The results are for ecosystems but suggest similar values to the individual case studies for forest land. Average opportunity costs at a global level are estimated at US\$85/ha/year for 5% of land, rising to US\$90/ha/year if 15% of land was withdrawn from production. At a regional level, the authors estimate opportunity costs at around US\$30/ha/year in Australia, New Zealand, Canada, and most of the developing world, and US\$100-200/ha/year in Southeast Asia and the USA. Estimated opportunity costs were significantly higher in Europe, at US\$1200/ha/year, and reached over US\$6000/ha/year in Japan.

Table 8 – Opportunity costs of forest biodiversity conservation

Reference	Location	Opportunity costs (US\$/ha/year)
Lewandrowski <i>et al</i> (1999)	Global	Reduction in GDP per ha, assuming 5% or 15% of land withdrawn from production: World: \$85.23/89.83 USA: \$180.63/189.77 Canada: \$31.37/32.95 EC: \$1211.12/1276.40 Japan: \$6187.02/6501.68 Other East Asia: \$62.86/66.76 Southeast Asia: \$100.39/106.67 Australia and New Zealand: \$28.85/30.16 Rest of world: \$35.17/36.08
van Kooten and Sohngen (2007)	Global	Mean cost of carbon sequestration projects = \$1612/ha in 2005\$; Range from \$6.63/ha - \$20606/ha
Bruner <i>et al</i> (2003)	Global	Costs of land purchase for expanding protected areas to cover most immediate priority areas for biodiversity conservation: up to \$5.2 billion per year, equivalent annual value (based on average costs of previous purchases – future purchases likely to be lower)
Grieg-Gran (2006)	Various	Opportunity costs of foregone land use plus plausible timber harvesting (annual equivalent values): Cameroon: \$38/ha/year DRC: \$39/ha/year Ghana: \$47/ha/year Bolivia: \$41/ha/year Brazil: \$24/ha/year PNG: \$89/ha/year Indonesian: \$49/ha/year Malaysia: \$58/ha/year
Carret and Loyer (2003)	Madagascar	Opportunity costs of protected areas: \$1.7 per ha per year (although increasing to \$5.52 per ha per year by the 15 th year – based on loss of income for forest households)
Ferraro (2002)	Madagascar	Opportunity costs for local households: \$28.87/ha/year

⁶ Roughly 12% of the earth's land area is currently under some form of legal protection, although this includes protected areas in which agriculture is permitted, as well as areas subject to illicit farming and other illegal activities (Chape *et al*, 2005).

Kramer et al (1995)	Madagascar	Opportunity costs for local households: \$7.3-8.7/ha/year
Emerton (1999)	Kenya	Opportunity costs for local households of not converting to agriculture: \$368/ha/year
Howard (1995)	Uganda	Average gross margins for cultivated land: \$128/ha/year Average gross margins for livestock production systems: \$8.1/ha/year Total opportunity costs of protected areas: \$36.1/ha/year – unevenly distributed between regions.
Naidoo and Adamowicz (2005)	Uganda	Have costs of conservation for each number of bird species (and benefits) – only present as graph, so not sure of exact figures.
Chomitz et al (2005)	Brazil	Median value of land: \$16.12/ha/year (2000 US\$) Cheapest 10,000ha of land in the zones with 'high forest cover' have average price of \$5.88/ha/year
Naidoo and Ricketts (2006)	Paraguay	Heterogeneous costs, ranging from \$0 to \$41/ha (annual equivalent value). Varied by land tenure, slope, soil type and location. Average opportunity costs across whole reserve: \$2.7/ha (annual equivalent value).
Butry and Pattanayak (2001)	Indonesia	Total losses to logging households: \$24.23/ha/year
Wilson et al (2007)	Malaysia and Indonesia	Sumatra: US\$0.86/ha Borneo: US\$0.99/ha Sulawesi: US\$0.69/ha Java/Bali: US\$7.06/ha Southern peninsular Malaysia: US\$24.81/ha
Bui Dung The and Hong Bich Ngoc (2006)	Vietnam	Average WTA to adopt sustainable management practices: \$9/ha/year Estimated transactions costs: \$18/ha
Ninan and Sathyapalan (2005)	India	Opportunity costs of not converting land to coffee production: Annual equivalent value - \$251-489/ha Costs of wildlife damage for coffee growers: \$26.5/ha/year
Strange et al (2006)	Denmark	Cost of preserving 740 of 763 priority species is \$412-638/ha/year - based on very approximate estimate of land area.
Horne (2006)	Finland	Forest owner WTA for conservation of small areas of forest: \$223/ha/year Forest owner WTA for larger areas of forest: \$398/ha/year
Siikamaki and Layton (2007)	Finland	Opportunity costs per site: median=\$738; mean=\$6,861
Sinden (2004)	Australia	If policy requires farmers to retain 30% of farm as native vegetation: \$4.4/ha/year if would choose to maintain 15% native vegetation. \$7.47/ha/year if would choose to maintain 5% native vegetation.
Shaikh et al (2007)	Canada	WTA compensation for converting to agroforestry: \$2.66-\$21.98/ha/year if account for uncertainty; \$11.4/ha/year if assume respondent certainty
Polasky et al (2001)	USA	Costs of species conservation/ha/year (site constrained/budget constrained) 350 species: \$1,220 / <\$122. 400 species: \$4,272 / \$220 415 species: \$7,300 / \$3,396
Ando et al (1998)	USA	Minimum cost of conserving habitats of 453 species: \$72/ha/year (AEV) - very approximate Minimum cost of conserving habitats of all 911 species: \$612/ha/year (AEV) - very approximate
Huang and Kronrad (2001)	USA	Necessary compensation ranges from \$.82/ha/year to \$71/ha/year - higher amounts are required when discount rates are assumed to be higher.

2.3.2 External costs

In certain cases, the conservation of forest biodiversity imposes external costs on people living nearby. The most common example is where wild animals in protected areas cause damage to neighboring crops or livestock or, in some cases, threaten human safety. In broad terms, these costs are relatively small: Emerton (1999) estimates the costs of crop damage by wild animals from the Mount Kenya Forest Reserve at US\$5/ha/year, while Madhusudan (2003) estimates the costs of damages by animals from the Bhadra Tiger Reserve in India at US\$0.83/ha/year for livestock losses and US\$1.58/ha/year for crop damage. However, for individual households these damages can be significant. In the Indian case study, for example, households that experienced damages lost an average of 11% of their annual crop production or 14% of their livestock assets.

Table 9 – External costs of biodiversity conservation

Reference	Location	External costs (US\$/ha/year)
Emerton (1999)	Kenya, Mount Kenya Forest Reserve	Damage to crops by wild animals: \$5/ha/year
Madhusudan (2003)	India, Bhadra Tiger Reserve, Karnataka, South India	Average value of livestock losses from large carnivores: \$0.83/ha/year. Average value of crop damage by elephants: \$1.58/ha/year.

2.3.3 Management/implementation costs

Grieg-Gran (2006) reviews estimates of management costs from programmes to avoid deforestation in a range of tropical countries. These range from US\$3.5-13/ha/year and are broadly consistent with other studies reporting management costs of protected areas in developing countries. The European Natura 2000 programme, which involves networks of nature reserves in various ecosystems, was reported to have considerably higher management costs of US\$56-94/ha/year (Markland, 2002), possibly reflecting higher labor costs and more intensive management.

Cullen *et al* (2005) examine biodiversity conservation programmes in New Zealand and report that projects carried out on smaller areas of land exhibit significantly higher costs. This reflects the fact that some management costs are fixed, regardless of the scale of the scheme. They also demonstrate that accessibility is important, as conservation projects on offshore islands cost on average over 10 times more than similar projects on the “mainland” islands of New Zealand.

Table 10 – Management costs of biodiversity conservation

Reference	Location	Management costs (US\$/ha/year)
Bruner <i>et al</i> (2003)	Global	Management costs of expanding protected areas to cover most immediate priority areas for biodiversity conservation: \$2.1 billion per year, or \$5.25/ha/year (based on conservation of 4 million km ²).
Carret and Loyer (2003)	Madagascar	Costs of network management: \$4.70 per ha per year (based on estimates of management organisation)
Cavatassi (2004)	India, Nepal and Philippines	Maharastra, forest rehabilitation: \$17.1/ha/year
Emerton (1999)	Kenya	Direct forest management costs: \$1.5/ha/year

Grieg-Gran (2006)	Various	Administration costs associated with payment schemes \$3.5-13/ha/year. Administration costs expected to rise significantly each year if reduction in deforestation maintained so that \$35-143/ha/year by year 10. Monitoring costs estimated at least \$2.3/ha/year.
Howard (1995) Markland (2002)	Uganda EU	Management costs of protected areas: \$4.8/ha/year Management costs of Natura 2000 network: \$56-94/ha/year

2.3.4 Expenditure on Biodiversity Conservation

An alternative to measuring or predicting the individual components of the total cost of forest biodiversity conservation involves calculating the actual expenditure under existing programmes. Note that conservation expenditure could be seen as a manifestation of the benefits of forest biodiversity, as it is evidence of willingness to pay for conservation. While actual spending on biodiversity conservation should not be interpreted as an average value for WTP, because of the significant scope for free riding, it does provide a lower bound estimate of benefits. Within our framework, such expenditures are treated as costs as they represent the actual costs that are incurred by conservation organisations to achieve current levels of conservation.

Actual expenditure on conservation is very often significantly less than the true economic cost. This is because reported spending by conservation organizations typically includes only the costs directly incurred by government or other institutions that fund a particular programme. Opportunity costs and external costs are often borne by other groups. For example, a protected area may be established without compensating the people who, as a result, are denied alternative uses of the land. In such cases, direct conservation expenditure may be much lower than the opportunity cost of protecting the land. Even where attempts are made to compensate for opportunity costs, precise targeting may not be possible, with the result that some land users are overcompensated. Part of the expenditure would then represent (inefficient) transfers between groups rather than the true economic costs of conservation.

Most published information on conservation spending, particularly at the global or regional level, does not distinguish between spending on forest biodiversity and spending to conserve other ecosystems. However, it is possible to obtain information on expenditure on forest conservation at programme-level. Proano (2005) examines three forest conservation projects in Ecuador, which are primarily aimed at watershed protection. These projects involve expenditure of between US\$3-22/ha/year, mainly on payments to farmers in return for changes in their land use practices. Rojas and Aylward (2003) study expenditure on the Payment for Environmental Services programme in Costa Rica, where funds are collected from a range of private companies, international donors and visitors to national parks to support payments to farmers of US\$38/ha/year for conservation of forest land and US\$94/ha/year for reforestation.

On the global scale, Bruner *et al* (2003) estimate total expenditure on protected areas at around US\$8 billion per year, of which approximately 60% covers forested land. At a regional level, Castro and Locker (2000) survey a range of donor institutions and report that 65 donors spent \$3.26 billion between 1990 and 1997 on conservation projects in

Latin America and the Caribbean, of which 66% in forest areas. The bulk of this funding came from multilateral institutions, such as the World Bank and the Inter-American Development Bank, and official donors such as the US and Canadian agencies for international development.

Some studies report conservation spending on a per hectare basis for different locations. James *et al* (1999) found that spending on biodiversity conservation was less than US\$1/ha/year, on average, in North Africa and Middle East, developing countries in Asia, and Russia and the CIS. Expenditure ranged between US\$1-3/ha/year in Sub-Saharan Africa, Latin America and the Caribbean, and Australia and New Zealand. In North America, average spending was higher at US\$9/ha/year. Conservation in Europe and the Pacific was relatively costly, at US\$21/ha and US\$28/ha respectively, while in developed East Asia, costs reached up to US\$131/ha/year. This variation in cost reflects differences in opportunity costs, which in turn are affected by the level of development of the country in which the conservation takes place, and the amount of available land.

The study by James *et al* (1999) provides average levels of expenditure across large areas. However, the costs of individual projects vary much more widely. Van Kooten and Sohngen (2007) carry out a meta-analysis of 68 studies estimating the costs of individual projects aimed at increasing afforestation or conserving existing forests. The costs of planting and maintaining forest land range from as little as US\$7/ha/year up to US\$21,000/ha/year, with an average of US\$1,600/ha/year.

Table 11 – Expenditure on biodiversity conservation

Author	Geographical area	Costs (US\$/ha/year)
Balmford et al (2003)	Global	Wilderness areas e.g. Gobi Desert, Himalayas, Amazon: <\$0.1-6/ha/year - typically around \$2/ha/year More densely settled areas of Latin and Central America, Africa and Asia: \$12->470/ha/year - typically around \$100/ha/year Developed countries: \$470-3,800/ha/year in US, \$1.5-4,700/ha/year in UK.
Bruner et al (2003)	Global	Current spending: approximately 8 billion per year. Total spending required to cover priority areas: \$22 billion per year for 10 years
Castro and Locker (2000)	Latin America and Caribbean	65 donors spent \$3.26bn on 3489 conservation projects between 1990 and 1997. 66% of funding to tropical and sub-tropical broadleaved forests; 29% to grasslands, savannahs and dry shrublands (i.e. not forests); 3.61% to mangroves; and .09% to conifer and temperate broadleaved forests.
James et al (1999)	Global	Global expenditure: \$4.84/ha/year North America: \$9.08/ha/year Latin America and Caribbean: \$1.09/ha/year Europe: \$20.67/ha/year Russia and CIS: \$0.83/ha/year Developed East Asia: \$131.07/ha/year Developing Asia: \$0.71/ha/year Sub-Saharan Africa: \$1.26/ha/year North Africa and Middle East: \$0.44/ha/year Australia and New Zealand: \$2.86/ha/year Pacific: \$28.44/ha/year

Lapham and Livermore (2003)	Global, mainly tropical	European Commission: estimated \$189million/year spent on biodiversity in developing countries. France: \$12.1-13.1m/year Germany: \$56.7million/year GEF: expected to spend \$960million on biodiversity between 2003 and 2006. Japan: \$3.6billion in Environmental ODA in 2000. Also \$25million over 5 years for grants to NGOs and community groups. ⁷ The Netherlands: annual budget for environment-sector development assistance is \$105.9million, but this includes contributions to multilateral institutions. UK: spending on biodiversity activities by DFID estimated at \$31.2-52million/year USA: \$128million/year on biodiversity and endangered species.
Molnar et al (2004)	Global	Estimated spending on protected areas: \$1.4/ha overall, and \$5.5/ha in global hotspots.
Wilson <i>et al</i> (2007)	Global (South Africa, Chile, Australia and California)	All values annual equivalent value. Australia: Control of invasive predators - \$3.3/ha; Management of fungal infestations - \$232/ha; Reversing habitat fragmentation - \$136/ha Chile: Invasive plant control - \$57/ha; Fire suppression - \$0.24/ha; Land acquisition for reserves - \$126/ha South Africa: Avoid agricultural conversion - \$239/ha; Avoid urban development - \$239/ha; Invasive plant control - \$420/ha California: Avoid urban development - \$457; Fire management - \$405/ha; Invasive plant control - \$1491/ha; Watershed management for invasive riparian plant control - \$2009/ha; Avoid agricultural expansion - \$458/ha
Cullen et al (2005)	New Zealand	Offshore islands - \$453.18 Mainland islands - \$32.29
Richie and Holmes (2001)	USA	Total state level expenditure on wildlife diversity programmes: \$134,898,266 Range of expenditure across states: \$50,000 - \$24.3 million
Proano (2005)	Ecuador	FONAG - the Water Fund for Quito, watershed protection projects: \$3.3/ha/year Cuenca Water Fee for protected area: \$22.15/ha/year Pimampiro water sources, avoiding deforestation: \$5.4-11/ha/year depending on type of land (primary forest, secondary forest, farmland).
Rojas and Aylward (2003)	Costa Rica	Much of the expenditure listed forms part of contribution to PES scheme, which pays farmers approximately \$38/ha/year for conservation and \$94/ha/year for reforestation.

2.3.5 Summary of the Costs of Conserving Forest Biodiversity

Case studies estimating the opportunity costs of protecting forest biodiversity broadly find costs below US\$100/ha/year, and often below US\$10/ha/year in most developing countries as well as in Australia, Canada, New Zealand and the USA. These results hold whether the land would otherwise be logged for timber or converted to agriculture, although where land would only be used for NTFP harvesting, opportunity costs tend to be lower. In Europe, on the other hand, the opportunity costs of conserving forest biodiversity are generally higher, between US\$500-1000/ha/year, while opportunity costs in Japan may be higher still.

Management costs range from US\$2-20/ha/year, although for small projects and protected areas on islands they may be higher. In addition, as with opportunity costs, management costs appear to be relatively high in Europe, with one study estimating them at US\$56-94/ha/year.

In addition to opportunity costs and management costs, there may be external costs, for example if wild animals in protected areas cause damage to crops or livestock belonging to local communities. These costs tend to be relatively small, on a per hectare basis, but can be significant for individual households.

Expenditure on programmes where private landowners receive payments in return for adopting environmentally-friendly land use practices provides an estimate of the costs of conserving forests on private or community-managed lands. In Ecuador, for example, farmers receive between US\$3-22/ha/year for land management changes, while in Costa Rica, farmers receive US\$38/ha/year for forest conservation or US\$94/ha/year for reforestation. By comparison, the available data for protected areas shows expenditure averaging less than US\$10/ha/year in most regions or about US\$21/ha/year in Europe. This suggests that opportunity costs are not fully compensated or, alternatively, that within each region, the areas of land with the lowest alternative values are currently under protection.

2.4 Overall Conclusions and Policy Implications of the Case Study Evidence

Valuation data on the costs and benefits of forest biodiversity conservation may be used to aid decision making, especially where trade-offs arise between conservation and other policy goals. More specifically, the case study data can be used for: cost-benefit analysis, ecosystem accounting, priority-setting for conservation policy, determining efficient levels of payment for ecosystem services, assessing damages to natural resources, and evaluating alternative policy options or scenarios. In this concluding section we consider what the case study data can say in relation to some of these decisions, the limitations of the data, and key gaps in the knowledge base.

2.4.1 Conclusions from the cost-benefit evidence

As noted above, one important use of case study data is to compare the costs and benefits of conservation, either at an aggregate level or for a particular area, in order to assess whether conserving forest biodiversity is worthwhile. An extension of the comparison of costs and benefits of biodiversity protection in a particular location is wider, spatial mapping of sites where benefits exceed costs or where conservation objectives may be met at the least cost.

Global lessons

In theory, the costs and benefits of forest biodiversity conservation could be assessed at a global level to ascertain whether there is a case for more, or perhaps less, funding for conservation. However, there are several reasons why this is not a productive use of the available case study evidence. First, most case studies aim to value selected components of total economic value; the results are not necessarily additive, as there may be trade-offs between different types of values. Second, the magnitude of the values is determined

by the quality of the ecosystems to which they relate and the scarcity of particular forest benefits relative to demand at a local or global level. Most case studies of forest benefits focus on areas with relatively high values, for example because they are visited by many tourists, they are important for NTFP collection, or because they provide habitat for endangered species. Hence a simple extrapolation of case study results to all forest land is likely to result in a significant over-estimation of benefits or costs or both. More useful than attempting to estimate total costs and benefits at a global level, is to look at alternative locations or types of forest to determine whether benefits exceed costs in specific cases, or to identify where the economic returns (net benefits) to conservation are likely to be highest.

Regional distribution of costs and benefits

Given the extent of variation in both costs and benefits by forest type and forest location, it is clear that there will be cases in which benefits are significantly higher than costs and also cases where the costs of conservation far exceed the benefits. As the benefits of forests for NTFP collection or for bioprospecting tend to be relatively small, there are unlikely to be many locations where they are sufficient, on their own, to justify the costs of conservation. In contrast, ecosystem services such as watershed protection and carbon sequestration can be very valuable in some contexts, while forests with high levels of biodiversity can also provide high non-use values and, in some cases, high recreational values. Of these different forest values, carbon sequestration and non-use values are global public goods and therefore do not depend on the size of the local population. On the other hand, watershed protection and recreational benefits are highly correlated with the number of people living in the vicinity and the relative accessibility of the site. This indicates that areas with low conservation costs, which are remote and have low population densities, will also tend to deliver lower local benefits from conservation. An exception would be if the forest was 'special' (i.e. having few substitutes) in some way and therefore had a high existence value.

The case studies reviewed above on the benefits of conservation suggest that, overall, the economic values of tropical forests are generally higher than the values of temperate forests. As tropical forests are most frequently located in regions where the costs of biodiversity conservation are relatively low, we would expect benefit-cost ratios to be higher, on average, for the conservation of biodiversity in tropical forests. Naidoo and Iwamura (2007) develop a global map of the opportunity costs of conservation, based on flows of benefits from agriculture. They find that values of land for agriculture are highest in North America, Europe, India and Southeast Asia, and lower in much of Africa, South America, Australasia and the Pacific. When regional costs are overlaid with biodiversity hotspots, the latter regions are found to contain the most cost-effective locations for biodiversity conservation. This study does not account for timber production as an alternative use of land, nor does it account for the risks of biodiversity loss in different regions, but it does show how spatial comparison of costs and benefits can provide useful information for prioritisation of conservation investments.

Site specific lessons

Using the results of the case studies discussed above to draw *wider* conclusions about the relative costs and benefits of conserving forest biodiversity results in statements that are, inevitably, rather general. In order to assess properly the net benefits of conserving forest biodiversity, the most useful studies look at both the costs and benefits of conservation in a particular location. Unfortunately, such studies are in short supply, although there are several policy conclusions that can be drawn from the available data.

A first conclusion is that, in cases where those who gain and lose from a particular conservation programme are located in the same region, with similar levels of income, comparisons between costs and benefits are relatively straightforward and a strong case can be made for adopting a conservation policy when benefits exceed costs. For example, Kniivila *et al* (2002) compare the costs and benefits of conserving an additional 5% of forest land in Finland and find that the stated benefits of local residents are nearly five times greater than the opportunity costs, in terms of timber harvests foregone. Gong (2004) uses a similar methodology to assess an endangered species programme in Guizhou Province in China, and finds that the benefits of conservation are nearly ten times greater than the estimated opportunity costs. Such studies can provide strong support for domestic conservation policy.

In contrast, the greater the distance in space, time or economic status between those incurring costs and those benefiting from biodiversity conservation, the more uncertainty there is in estimating net benefits and the weaker the case for conservation. For example, Kramer *et al* (1995) estimate the opportunity costs of establishing a national park in Madagascar at US\$500,000-700,000, while the benefits are estimated at US\$800,000-2,160,000 (both expressed in term of NPV). However, the opportunity costs are borne mainly by relatively poor, local villagers, while the benefits are enjoyed mainly by relatively rich, foreign tourists. The study shows that if the benefits to tourists were captured, for example through park entry fees, they would be sufficient to compensate for the losses incurred by local people. However, if compensation did not take place, the park would have negative distributional consequences, despite the positive aggregate welfare effects. The relevance of uncompensated opportunity costs is also highlighted by Emerton (1999), who estimates the costs and benefits of a protected area in Kenya. This study shows that the local opportunity costs of conservation significantly exceed local benefits, and concludes that local communities will only have incentives to conserve forest areas if global benefits can be captured and used to win their support.

Most studies of individual conservation programmes assume that the features and implementation of the programme are fixed. However, a second important conclusion from the review of case studies is that, in order to maximise the efficiency of policy interventions, it is essential to consider how costs and benefits vary under alternative programme designs. Several studies examine the extent to which forest land should be protected from economic activity, or the locations in which conservation should take place, in order to maximise net benefits. Naidoo and Adamowicz (2005), for example, compare the opportunity costs of increased protection of forest land to the increased tourism revenues that result from higher levels of biodiversity in forest reserves in Uganda. They find that the marginal benefits of increased biodiversity exceed marginal

costs, up to the point where 80-90% of bird species in the reserves are protected, while for additional protection (beyond 90%), costs generally exceed benefits. Similarly, Naidoo and Ricketts (2006) carry out a spatially explicit cost-benefit analysis of conservation in a forest reserve in eastern Paraguay; they find that if all benefits apart from carbon storage are considered, only the core of the reserve, where no forest conversion has taken place, is worth conserving, whereas if carbon storage is included, then other patches of forest also provide positive net benefits under conservation.

Such studies show that the net benefits of conservation depend on the way that it is implemented, in particular the degree of protection and its location. Detailed spatial data on conservation costs and benefits is thus extremely useful for maximising net benefits, but is also data-intensive.

2.4.2 Policy priority setting

The case study data on the costs and benefits of conserving forest biodiversity can aid priority setting for policy intervention in various ways. One approach is to identify the locations where, and the extent to which, the benefits of conservation exceed the costs, as discussed above. Another contribution is to identify the relative importance of different types of value associated with biodiversity conservation.

This contribution is twofold: first, it allows us to assess whether the most significant economic values of conservation are accounted for in land use decisions, within existing market and non-market institutions. If they are not, there is potential for significant improvements in welfare through a better alignment of values and decision-making, as well as a high likelihood of losses from policy inaction.

A second contribution is to indicate which values would produce the greatest opportunities for conservation if they were captured by those responsible for the land use decisions. This is most relevant for market creation policies, which develop the necessary institutions and skills to enable forest landowners to be compensated financially for providing benefits that people are willing to pay for through the market. However, it is also relevant to government decisions about domestic forest policy, in cases where foreign citizens have preferences over the outcomes.

Identifying neglected values

The case study results suggest that the value of conserving forest biodiversity for bioprospecting is unlikely to be very high in a majority of cases. Moreover, in those cases where bioprospecting values are high, the private sector will have strong incentives to develop measures to encourage conservation. Forest NTFP values are also relatively low and will usually be captured privately by local communities using forest resources.

Whether this means that NTFP values are accounted for in decision making depends on the nature of property rights to forest land; if the individuals harvesting NTFPs also hold secure rights to log for timber or convert to agriculture, we might expect that NTFP

benefits would be fully accounted for. However, where the rights to log, convert or otherwise disturb forest land are held by other parties, for example through government concessions, these local benefits are unlikely to be given much weight in land use decisions.

Benefits from tourism are relatively high but, as with other direct use values, they can often be captured with minimal policy intervention. This is primarily because recreational benefits have some of the characteristics of private goods and, as such, can provide compensation for conservation through the market. Private demand for tourism in diverse forest ecosystems can provide adequate incentives for their protection, at least where tourist demand exceeds the opportunity costs of the land for agriculture or other uses. Nevertheless, public intervention may be required where local communities lack the necessary skills and experience to market ecotourism successfully. Alternatively, where public provision is deemed more suitable for the provision of recreation or tourism benefits, this could still be financed through park entry fees and/or concession fees.

Carbon sequestration is another example of a forest value that is relatively high but which may be captured through existing market structures without the need for special biodiversity policy. At present, carbon benefits are not often considered in land use decision making, but growing attention to combating climate change has resulted in the development of new mechanisms to compensate landowners for forest conservation or reforestation, and this is likely to continue. The important policy issue in relation to forest biodiversity is how to ensure that, where land is maintained under forest cover for carbon sequestration purposes, the biodiversity value of those forests is also accounted for. This may be achieved through encouragement of forest conservation over reforestation, or by supplementing payments for carbon sequestration with payments for biodiversity.

In contrast to the forest values discussed above, the case study evidence suggests that non-use (existence and bequest) values are not only generally high but also that there are only limited means by which these values can be captured at present. Eco-labelling programmes for “ecological” goods and services provide one mechanism through which some non-use values can be expressed through the market, while charitable donations are another way for those who hold non-use values for forest biodiversity to encourage its conservation. However, both of these funding mechanisms are subject to significant free-riding, which leaves only direct funding by domestic or international governments as the main vehicle for capturing non-use values.

The implication is that public intervention is most needed to ensure the conservation of forests with high non-use values, including increased financial flows. The latter is particularly important to ensure an adequate flow of funds from developed countries, where citizens express high non-use values for tropical forest biodiversity, to developing countries, where forest biodiversity is often high but incomes are low.

Forest valuation and incentives

The high values of forests for carbon sequestration, and for recreation and tourism, suggest that efforts to develop these markets could potentially generate sufficient income

from conserving forests to outweigh the income from deforestation, in many locations. Improvements in markets for NTFPs, on the other hand, may not provide forest-based incomes that exceed opportunity costs, apart from a few special cases where returns are unusually high or opportunity costs are very low. While there is considerable uncertainty about the benefits of conserving forests for bioprospecting, the available evidence suggests that they will generally not be sufficient to ensure conservation on their own. Whether markets for watershed protection or other ecosystem services are worth developing will vary considerably by location, which suggests that the current pattern of ad-hoc payments at local levels is appropriate. Finally, the case study evidence suggests that the non-use values of forest biodiversity can be extremely high and we would expect these values to have significant influence on forest land use decisions, if mechanisms existed to capture them. However, the scope for free riding means that non-use values can be only partially captured through voluntary market mechanisms, hence public intervention is needed to ensure that such values are, in fact, accounted for.

In conclusion, the case studies reviewed here suggest that focusing conservation policy on the non-use values of forest biodiversity could deliver large benefits, while the public good nature of these benefits means that, without direct government intervention, they are unlikely to be captured. Furthermore, non-use values are most likely to be large where opportunity costs are low, hence the required finance for conservation need not be very costly. Existing market and policy frameworks also mean that the benefits of forests for recreation and for carbon sequestration are increasingly accounted for in land use decisions. However, further development of these mechanisms could offer significant potential to reinforce the conservation of forest land, particularly if links with biodiversity are strengthened. Finally, although other direct use values, such as NTFP harvests and bioprospecting, may be important for conservation in some situations, they will generally not generate sufficient revenue to support forest conservation on their own.

2.4.3 The extent of the evidence and remaining gaps

In terms of the different types of economic benefit provided by diverse forests, the case studies reviewed in this report provide relatively good information on the values of direct uses such as NTFP harvesting, recreational activities and, to a lesser extent, bioprospecting. Data on the recreational values of forest land are most comprehensive and can be related to different forest types in different locations. In the case of temperate forests, it is also possible to compare preferences for more diverse with less diverse forests, while for tropical forests there is less information on the value of increased biodiversity relative to simple protection of forest land. We found little information on the value of NTFPs in developed countries, other than in the Mediterranean region, but this may simply reflect their relatively low economic importance in these countries. Exceptions are hunting and angling, both of which have high recreational value.

Data on the indirect use values of forests are less comprehensive, particularly for temperate forests. The available studies on the value of watershed protection tend to relate to very specific cases, and are difficult to generalise to a wider context. In contrast,

because the benefits of carbon sequestration are not determined by the location of a forest, estimates of value can be assigned to many forest types. On the other hand, these estimates are subject to considerable uncertainty, as they are dependent on assumptions about the future damage costs of climate change.

Many studies have been carried out on the non-use values of forests, in particular the values that residents of the USA or Europe place on forests in their own country or state, and values that residents of developed countries place on tropical forests in developing countries. Less is known about the values that citizens of developing countries place on forest diversity in their own or other countries. Non-use values seem to vary significantly, depending on the number of people over which they are aggregated. In order to use the results of these studies more generally, it is necessary to understand how non-use values vary across different populations.

Overall, the case studies on the benefits of conserving forest biodiversity (or forest resources) allow us to distinguish different types of economic value, and to a certain extent different types of forest ecosystem and different geographical locations. Most of the case studies value discrete changes in biodiversity or forest resources, or average values for existing forest stocks, rather than marginal changes. A small number of studies quantify the marginal recreational or existence value of an additional species within a forest, but more generally there is little information available on how welfare changes with marginal increases in either forest area or forest diversity.

Another key gap in the benefits data relates to the relative value of greater forest diversity compared to increased forest area. For the specific cases of recreational values and non-use values, some studies assess the extent to which households value increased diversity, largely in temperate forests. However, more generally it is not clear how the economic benefits of forest conservation relate to their diversity. Finally, little is known about the relative value of alternative spatial patterns of conservation, or the differences in preferences for marginal increases in diversity compared to avoided losses of diversity.

On the cost side, the opportunity costs of conserving forest biodiversity, based on either land values or modelling of the expected returns to alternative uses of land, are fairly well documented in terms of average values across broad areas. Data on expenditure on biodiversity conservation across regions is also available, although as this relates to all ecosystems, it is not clear whether the pattern of spending is the same for forest biodiversity alone. Key areas in which further research is required include the costs of implementation for alternative policy mechanisms, and the returns to different levels of expenditure. The available information on costs can provide rough approximations for assessing the net benefits of biodiversity conservation in different regions, but is less useful for comparing alternative conservation methods or for assessing more precisely which particular locations should be conserved.

Additional information covering the gaps discussed above would significantly improve the comparison of conservation benefits and costs. For example, more robust data on the marginal value of changes in biodiversity, combined with marginal rather than average

estimates of the costs of protection, would help to ensure more efficient allocation of scarce conservation funds. Moreover, given that many of the costs and benefits of protecting forest biodiversity depend on the location and other specific characteristics of the forest, it would also be helpful if more case studies analysed both benefits and costs at the site level. One promising avenue is spatial cost-benefit analysis, which goes beyond simple comparison of the total costs and benefits of specific locations or programmes, to assess the optimal spatial pattern of conservation.

2.4.4 Limitations of the case study data

There is wide variation in the values measured by the case studies reviewed here, as well as the methods used for measurement. As an illustration, Mogas *et al* (2006) value the additional recreational benefits of increasing forest cover in Catalonia from 40% to 50% of total area, while Mill *et al* (2007) estimate willingness to pay for recreation in natural forests relative to commercially managed forests in Ireland. Both studies value forest recreation, but they do not allow us to compare the values of forest recreation in Spain with values in Ireland, as the former study values changes in forest quantity while the other values changes in forest quality.

Other important sources of variation include: whether a study estimates the marginal value of changes in forest resources or the total benefits arising from existing stocks; the estimation methods used, including payment mechanism and payment frequency; and sample characteristics. Where multiple studies share similar objectives and methods, they can provide an indication of the relative magnitude and determinants of particular values, and thus allow comparisons between, for example, different locations or different forest types. However, where there are many differences between studies or large differences in what is being measured, then the direct comparison of values may be less meaningful. More generally, variation in study design and econometric methods makes valuation studies less comparable than other sources of economic data (e.g. field experiments etc). This variability also undermines meta-analysis and benefit transfer studies.

Another important limitation arises from the non-linear relationship between changes in biodiversity and benefit flows. The most useful case studies for making policy decisions in the context of non-market goods and services are those that estimate the change in the value of benefits arising from a given change in the amount or quality of those goods or services. However, if we are valuing biodiversity, then as the diversity of a forest changes, the benefits arising from it may remain constant until a certain threshold is reached, at which point large changes may occur. In extreme cases, there may be no obvious change in the benefits arising from an ecosystem as it becomes less diverse, until the point of no return is passed and the ecosystem simply collapses. One way to deal with this possibility is to estimate the insurance value of maintaining forest ecosystem, based on changes in probability of ecosystem collapse. However, this is not done in any of the available case studies, probably due to lack of data on where ecological thresholds lie.

Another partial solution is to value the benefits of the existence of forests, or of certain types of forests, rather than the diversity of those forests *per se*. In practice, this is what the majority of case studies do. For certain types of value, there is an approximately linear relationship between the forest area and the benefits obtained, making it possible to estimate the value of marginal changes in forest land. This is the case for carbon sequestration benefits and, to a lesser extent, for NTFP harvesting. However, other types of value, such as the recreational value of forest land or the benefits of watershed protection, may have little or no relationship to the area of the forest, and will therefore not change as forest area changes. Non-use values are also likely to relate to the existence of *some* forest of a certain type, rather than the total amount of forest land.

In such cases, it may not be meaningful to estimate marginal values per hectare of forest. Alternative denominators may be required, relating to the numbers of people affected by forest changes or the number of forest units or particular habitats within a region. However, a disadvantage of such approaches is that, if different denominators are used for different types of value, they cannot then be compared with each other.

On the cost side, it is not possible to assess or compare the costs of conserving forest biodiversity unless there is a clear metric for what is being conserved, and how much. There are useful indicators of biodiversity or ecosystem changes, but these have so far not been related to the costs of conservation. Equally, costs will vary according to the conservation mechanism used.

Finally, the marginal costs of conservation are not constant because they are primarily driven by the opportunity cost of land. As this is determined by land scarcity, then opportunity costs will tend to increase as more land is conserved. This means that if land is dedicated to conservation on the basis of the lowest cost land first (assuming constant benefits for simplicity), then marginal costs may initially be very low. However, studies of species conservation have shown that attempts to conserve all species or all habitats within a certain region can result in very high costs for the last units conserved (e.g. Ando *et al*, 1998, Naidoo and Abramovicz, 2005). This means that the costs of conservation can only be estimated in relation to the amount of conservation that is desired.

Overall, the case study findings provide important evidence on the approximate magnitude of the benefits and costs of forest biodiversity conservation, and suggest where benefits are most likely to exceed costs. There are also studies that illustrate how net benefits could be estimated at a local scale, which allows us to assess whether investments in biodiversity conservation should be made at particular sites, and how conservation objectives could be achieved most cost-effectively. However, the issues raised above show that caution should be used when attempting to aggregate and extrapolate the case study results on a large scale.

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