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Assessing the Economic Value of Ecosystem Conservation

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and IUCN—The World Conservation Union

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Executive summary

Valuation studies have considerably increased our knowledge of the value of ecosystems. Their usefulness has often been undermined, however, by a failure to properly frame them so as to address the specific question of interest. Unfortunately, environmental advocates in the media, government, business, and civil society have often seized on impressive but sometimes unsound valuation results and used them indiscriminately, and often inappropriately.

Valuation is not a single activity, and the seemingly simple question ‘how valuable is an ecosystem?’ can be interpreted in many different ways. It could be interpreted as asking about the value of the current flow of benefits provided by that ecosystem, for example, or about the value of future flows of benefits. It could also be asking about the value of conserving that ecosystem rather than converting it to some other use. These interpretations of the question are often treated as being synonymous, but they are in fact very different questions, and the answer to one will not be correct as an answer to the other.

This paper seeks to clarify how valuation should be conducted to answer specific policy questions. In particular, it looks at how valuation should be used to examine four distinct aspects of the value of ecosystems:

- **Determining the value of the total flow of benefits from ecosystems.** This question typically arises in a ‘national accounts’ context: How much are ecosystems contributing to economic activity? It is most often asked at the national level, but can also be asked at the global, regional, or local level.
- **Determining the net benefits of interventions that alter ecosystem conditions.** This question typically arises in a project or policy context: Would the benefits of a given conservation investment, regulation, or incentive justify its costs? It differs fundamentally from the pre-

vious question in that it asks about *changes* in flows of costs and benefits, rather than the sum total value of flows.

- **Examining how the costs and benefits of ecosystems are distributed.** Different stakeholder groups often perceive very different costs and benefits from ecosystems. Understanding the magnitude and mix of net benefits received by particular groups is important for two reasons. From a practical perspective, groups that stand to ‘lose’ from conservation may seek to undermine it. Understanding which groups are motivated to conserve or destroy an ecosystem, and why, can help to design more effective conservation approaches. From an equity perspective, the impact of conservation on particular groups such as the poor, or indigenous peoples, is also often of significant concern in and of itself.
- **Identifying potential financing sources for conservation.** Knowing that ecosystem services are valuable is of little use if it does not lead to real investments in conserving the natural ecosystems that provide them. Simply knowing that a protected area provides valuable watershed protection benefits, for example, does not pay the salaries of park rangers. Yet experience has shown that relying solely on government budget allocations or external donors for the necessary funding is risky. Valuation can help identify the beneficiaries of conservation and the magnitude of the benefits they receive, and thus help design mechanisms to capture some of these benefits and make them available for conservation.

These four approaches are closely linked and build on each other. They represent four different ways to look at similar data regarding the value of an ecosystem: its total value or contribution to society, the change in this value if a conservation action is undertaken, how this change affects different stakeholders—that is, who are the bene-

Approaches to valuation

| <i>Approach</i> | <i>Why do we do it?</i> | <i>How do we do it?</i> |
|---|--|--|
| Determining the total value of the current flow of benefits from an ecosystem | To understand the contribution that ecosystems make to society | Identify all mutually-compatible services provided; measure the quantity of each service provided; multiply by the value of each service |
| Determining the net benefits of an intervention that alters ecosystem conditions | To assess whether the intervention is economically worthwhile | Measure how the quantity of each service would <i>change</i> as a result of the intervention, as compared to their quantity without the intervention; multiply by the marginal value of each service |
| Examining how the costs and benefits of an ecosystem (or an intervention) are distributed | To identify winners and losers, for equity and practical reasons | Identify relevant stakeholder groups; determine which specific services they use and the value of those services to that group (or changes in values resulting from an intervention) |
| Identifying potential financing sources for conservation | To help make conservation financially sustainable | Identify groups that receive large benefit flows, from which funds could be extracted using various mechanisms |

beneficiaries and who are the losers—and how beneficiaries could be made to pay for the services they receive to ensure that the ecosystem is conserved and its services are sustained. Each of these approaches to valuation uses similar data. They use that data in very different ways, however, sometimes looking at all of it, sometimes at a subset, sometimes looking at a snapshot, and sometimes looking at changes over time. Each approach has its uses and its limitations. Understanding under what conditions one approach

should be used rather than another is critical: the answer obtained under one approach, no matter how well conducted, is generally meaningless when applied to problems that are better treated using another approach. In particular, using estimates of total flows to justify specific conservation decisions—although commonly done—is almost always wrong. Properly used, however, valuation can provide invaluable insights into conservation issues.

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Abbreviations

| | |
|------|-------------------------------------|
| ANS | Adjusted Net Savings |
| CBD | Convention on Biological Diversity |
| CV | Contingent valuation |
| GDP | Gross domestic product |
| GNI | Gross national income |
| IRR | Internal rate of return |
| MA | Millennium Ecosystem Assessment |
| NPV | Net present value |
| NTFP | Non-timber forest product |
| PA | Protected area |
| PES | Payments for environmental services |
| SMS | Safe minimum standard |
| TC | Travel cost |
| TEV | Total economic value |
| WTA | Willingness to accept |
| WTP | Willingness to pay |

1 Introduction

In the early 1990s a fascinating ecological experiment was conducted in the deserts of Arizona, USA. Dubbed ‘Biosphere 2’ (‘Biosphere 1’ is our planet), this project was an attempt to create a closed but self-sustaining artificial ecosystem that would provide a small group of people with all the food, air, water and other raw materials needed to survive indefinitely, with sunshine the only external input. While the experiment failed in one sense, given that the inhabitants were forced to abandon their artificial home (due to rising concentrations of CO₂ in the artificial atmosphere), many valuable lessons were learned. Among these was a new appreciation of the complexity of the natural processes that support life on earth.

The benefits provided by natural ecosystems are both widely recognized and poorly understood (Daily, 1997). What is increasingly clear, however, is that natural ecosystems are under enormous pressure around the world from the growing demands placed on them by human economies. Growth in human populations and prosperity translates into increased conversion of natural ecosystems to agricultural, industrial, or residential use, but also into increased demand for ecosystem inputs, such as fresh water, fiber, and soil fertility, as well as increased pressure on the capacity of natural ecosystems to assimilate our waste, including air and water pollution as well as solid waste. In short, we are asking more and more from natural ecosystems even as we reduce their capacity to meet our needs.

Stating that natural ecosystems and the services they provide are valuable immediately leads to the question: how valuable? This is an important question because other things are valuable as well. Maintaining ecosystems, whether through protected areas or through some other mechanism, requires expenditure of resources, and there are often many competing claims on these resources. Devoting more effort to conservation may mean

having fewer resources to address other pressing needs, such as improving education, health, or infrastructure. Conserving ecosystems and the goods and services they provide may also involve foregoing certain uses of these ecosystems, and the benefits that would have been derived from those uses. Not converting a forest ecosystem to agriculture, for example, preserves certain valuable ecosystem services that forests may provide better than farmland, but it also prevents us from enjoying the benefits that agricultural production can provide. To assess the consequences of different courses of action, it’s not enough to know that ecosystems are valuable, we also need to know how valuable they are, and how that value is affected by different forms of management.

It has often been argued that a major reason for our failure to conserve natural ecosystems is that we do not realize how valuable they are. The farmers deciding whether to burn a hectare of forest to clear it for agriculture focus on the potential crop yields they may obtain, but pay little attention to the many ecological services that would go up in smoke. Likewise, national ministers of finance often base their budget decisions solely on the basis of indicators such as GDP, foreign exchange balances, and tax receipts, in which ecosystem services either do not appear or are not recognized as such—indeed, perversely, GDP often identifies activities that destroy ecosystems as ‘benefits’. Not surprisingly, conservation budgets tend to get slighted.

Such concerns have led to an explosion of efforts to value natural ecosystems and the services they provide. The vast majority have focused on valuing only a sub-set of the benefits of particular ecosystems in specific locations (for example, the value of water filtration services provided by wetlands in Kampala, Uganda, see Emerton and others, 1998). Some more ambitious efforts have attempted to estimate the value of all services pro-

vided by broad categories of ecosystems (for example, the benefits of forests in Mediterranean countries, see Case Study 1 below), or even of all ecosystems on the planet (see Box 4.2 below).

Valuation studies have considerably increased our knowledge of the value of ecosystems, as well as of the strengths and limitations of different valuation methods. Another, less desirable outcome, however, has been growing confusion among decision-makers and non-economists about the validity and implications of ecosystem valuation. Unfortunately, environmental advocates in the media, government, business, and civil society have often seized on impressive but sometimes unsound valuation results and used them indiscriminately, and often inappropriately.

Valuation is not a single activity, and the seemingly simple question ‘how valuable is an ecosystem?’ can be interpreted in many different ways. It could be interpreted as asking about the value of the current flow of benefits provided by that ecosystem, for example, or about the value of future flows of benefits. It could also be asking about the value of conserving that ecosystem rather than converting it to some other use. These interpretations of the question are often treated as being synonymous, but they are in fact very different questions, and the answer to one will not be correct as an answer to the other.

Asking ‘how valuable is an ecosystem?’ also begs the question ‘how valuable to whom?’ The benefits provided by a given ecosystem often fall unequally across different groups. Ecosystem uses that seem highly valuable to one group may cause losses to another. Answering the question from the aggregate perspective of all groups (as is often the case in economic analysis), would thus give very different answers to answering it from the perspective of a particular group. Understanding the distribution of costs of benefits is also important when considering how to mobilize funds for conservation. Knowing that an ecosystem is valuable will not by itself ensure that it is conserved. Valuation can provide important insights into how conservation might be made financially sustainable—provided it is used the right way.

This paper seeks to clarify how valuation should be conducted to answer specific policy questions. In

particular, it looks at how valuation should be used to examine four distinct aspects of the value of ecosystems:

- **Determining the value of the total flow of benefits from ecosystems.** This question typically arises in a ‘national accounts’ context: How much are ecosystems contributing to economic activity? It is most often asked at the national level, but can also be asked at the global, regional, or local level.
- **Determining the net benefits of interventions that alter ecosystem conditions.** This question typically arises in a project or policy context: Would the benefits of a given conservation investment, regulation, or incentive justify its costs? It differs fundamentally from the previous question in that it asks about *changes* in flows of costs and benefits, rather than the sum total value of flows.
- **Examining how the costs and benefits of ecosystems are distributed.** Different stakeholder groups often perceive very different costs and benefits from ecosystems. Understanding the magnitude and mix of net benefits received by particular groups is important for two reasons. From a practical perspective, groups that stand to ‘lose’ from conservation may seek to undermine it. Understanding which groups are motivated to conserve or destroy an ecosystem, and why, can help to design more effective approaches to conservation. From an equity perspective, the impact of conservation on particular groups such as the poor, or indigenous peoples, is also often of significant concern in and of itself.
- **Identifying potential financing sources for conservation.** Knowing that ecosystem services are valuable is of little use if it does not lead to real investments in conserving the natural ecosystems that provide them. Simply knowing that a protected area provides valuable watershed protection benefits, for example, does not pay the salaries of park rangers. Yet experience has shown that relying solely on government budget allocations or external donors for the necessary funding is risky. Valuation can help identify the main beneficiaries of conservation and the magnitude of the benefits they receive, and thus help design mechanisms to capture some of these

benefits and make them available for conservation.

These four approaches are closely related, but distinct. As will be shown, they can be seen as looking at the same data from different perspectives. The specific answers to each of these questions can be very different, however, and the answer to one is often not meaningful when used as the answer to another.

The aim of this paper is not to provide detailed instructions on how to undertake valuation of ecosystem services, nor on how to use specific valuation techniques. There are many other sources that provide such instructions. Rather, the objective of this paper is to clarify how valuation can and should be used to address important policy questions that often arise—and how such valuation differs from that which would be undertaken to address a different policy question.

Chapter 2 begins by providing an overview of the conservation problems we are addressing. These go beyond the narrow focus on protected areas that has often characterized the debate, and also include other conservation efforts. Although protected areas have been and will continue to be important tools for conservation, many valuable ecosystem services are provided by other land uses, including agriculture and industrial forestry.

Chapter 3 provides a summary of the main valuation techniques, their applicability to different problems, and their strengths and limitations, as well as numerous references. Recent years have seen a substantial development of these techniques.

Chapters 4 to 7 discuss each of the four approaches to valuation outlined above in detail, showing how they differ and how they relate to each other, and providing guidelines on how to implement them and how to use the results to inform investment and policymaking.

Chapter 8 concludes by discussing the strengths and limitations of economic valuation. While economic

valuation—if used correctly—can provide useful information for policymaking on ecosystem conservation, it also has limitations.

Chapter 9 provides detailed case studies of the application of valuation in a number of contexts.

The view taken in this paper is that the purpose of valuation is to obtain reliable, objective information on the benefits and costs of conserving ecosystems so as to inform decisionmaking. In the context of evaluating a specific project or policy intervention, for example, it asks *whether* the resulting net benefits are sufficient to justify the costs of the intervention. This presumes that, in some cases, they may not be. All too often, valuation is used merely as a tool to provide ammunition in support of a predetermined position, with its results being discarded if they do not, in fact, support it.

We recognize that some people reject the assumptions and methods used to express environmental benefits in monetary terms. Although economic valuation methods are far from perfect, and are not the only way to assess ecosystem benefits, the view taken here is they are useful for illuminating trade-offs and guiding decisionmaking.

The focus on this paper is decidedly anthropocentric: the ecosystem benefits we consider are those that contribute to human welfare. This is not, of course, the only reason to be concerned about ecosystems. Many, drawing on a variety of ethical, philosophical, or cultural traditions, consider some ecosystems as having *intrinsic value*, irrespective of whether they contribute to human welfare (Goulder and Kennedy, 1997; Millennium Ecosystem Assessment, 2003:). There may be other reasons to conserve ecosystems besides the economic benefits they provide. Understanding the economic costs and benefits of using ecosystems is thus only one of many inputs that enter into decisionmaking. The concern of this paper is that such understanding should be as accurate, meaningful, and useful as possible.

2 The importance of ecosystem services

There is growing concern worldwide about the destruction and degradation of natural ecosystems and the attendant loss of biodiversity. On average, almost 15 million hectares of forest were lost every year during the 1990s, mostly in the tropics (FAO, 2001). 35 percent of mangrove forests have been lost in the last two decades (Valiela and others, 2001). An estimated 11 percent of the world's coral reefs have been lost, and an additional 16 percent severely damaged (Wilkinson, 2000). Managed ecosystems such as agricultural lands have also become increasingly degraded. These losses would once have been of concern only to biologists, but growing awareness of the importance of natural ecosystems and the goods and services they provide has made ecosystem degradation an important concern worldwide.

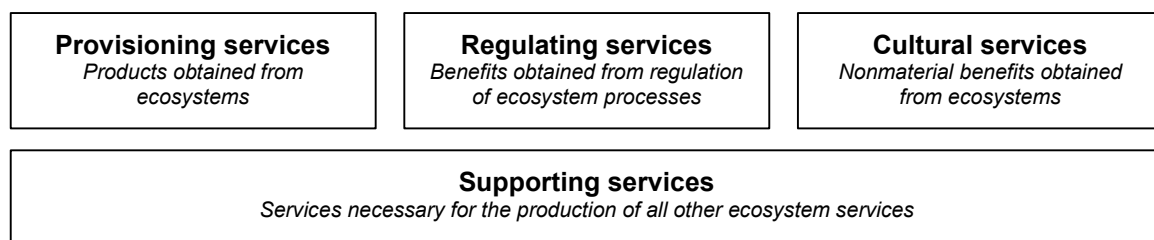
Ecosystems and the services they provide

Ecosystems, and biodiversity more generally, matter for many reasons. The reasons this paper focuses on are practical: ecosystems provide a wide variety of useful services that enhance human welfare. Without these services, we would be worse off in many ways. At the limit, we may not survive. But even degradation of ecosystem services falling well short of outright destruction would significantly affect our welfare.

The world's ecosystems provide a huge variety of goods and services. We are all familiar with the valuable commodities that natural ecosystems provide, such as edible plants and animals, medicinal products, and materials for construction or clothing. Many of us likewise value the aesthetic or cultural benefits provided by natural ecosystems, including beautiful views and recreational opportunities. What is less well known is the extent to which human economies depend upon natural ecosystems for a range of biological and chemical processes. Examples of ecosystem services include the purification of air and water; regulation of rainwater run-off and drought; waste assimilation and detoxification; soil formation and maintenance; control of pests and disease; plant pollination; seed dispersal and nutrient cycling; maintaining biodiversity for agriculture, pharmaceutical research and development and other industrial processes; protection from harmful ultraviolet radiation; climate stabilization (for example, though carbon sequestration); and moderating extremes of temperature, wind, and waves (after Daily, 1997).

We follow here the Millennium Ecosystem Assessment's (MA) definition of ecosystems as dynamic complexes of plant, animal, and micro-organism communities and the non-living envi-

Figure 2.1: Typologies of ecosystem services: the Millennium Ecosystem Assessment



Source: Millennium Ecosystem Assessment, 2003.

ronment, interacting as functional units. It is important to note that this includes managed ecosystems such as agricultural landscapes, and even urban areas. The MA classifies the services that ecosystems can provide into four broad categories: *provisioning services*, *regulating services*, *cultural services*, and *supporting services* (Figure 2.1). This typology separates services along functional lines. These categories illustrate the diverse ways in which ecosystems contribute to human welfare. Table 2.1 shows the main ecosystem types recognized by the MA and the principal services that each provides. Because these ecosystem ‘services’ are provided free of charge, as a gift of nature, their importance is often overlooked.

Despite the services they provide, natural ecosystems worldwide are under tremendous pressure. Forest ecosystems are being converted to other uses; wetlands are being drained; and coral reefs are being destroyed. Freshwater resources are increasingly modified through impoundment, redirection, extraction, land use changes that affect recharge and flow rates, and pollution. Agricultural soils and pasture lands are

being degraded from over-use. Some of these pressures are intentional effects of human activities, others are un-intended.

Approaches to conservation

The standard approach to conservation has been the establishment of protected areas (PAs). This approach cordons off certain areas and restricts their use. There has been considerable debate about the effectiveness of PAs as instruments for protection (Brandon and others, 1998). Recent research shows that PAs can be very effective in many cases (Bruner and others, 2001). However, their effectiveness is limited by the fact that many PAs are too small and isolated to sustain the full range of ecosystem services. Moreover, due to weak capacity and limited resources many PAs are little more than ‘paper parks’—protected in name only.

The limitations of PAs as a conservation strategy have led to increased attention being given to conservation efforts outside formally protected areas. Agricultural landscapes cover a large proportion of the world’s surface, for example.

Table 2.1: Main ecosystem types and their services

| <i>Ecosystem service</i> | <i>Ecosystem</i> | | | | | | | | | |
|---------------------------|-------------------|----------------|---------------|--------------|---------------------|----------------|---------------|--------------|-----------------|---------------|
| | <i>Cultivated</i> | <i>Dryland</i> | <i>Forest</i> | <i>Urban</i> | <i>Inland water</i> | <i>Coastal</i> | <i>Marine</i> | <i>Polar</i> | <i>Mountain</i> | <i>Island</i> |
| Freshwater | | | • | | • | • | | • | • | |
| Food | • | • | • | • | • | • | • | • | • | • |
| Timber, fuel, and fiber | • | | • | | | • | | | | |
| Novel products | • | • | • | | • | | • | | | |
| Biodiversity regulation | • | • | • | • | • | • | • | • | • | • |
| Nutrient cycling | • | • | • | | • | • | • | | | |
| Air quality and climate | • | • | • | • | • | • | • | • | • | • |
| Human health | | • | • | • | • | • | | | | |
| Detoxification | | • | • | • | • | • | • | | | |
| Natural hazard regulation | | | • | | • | • | | | • | |
| Cultural and amenity | • | • | • | • | • | • | • | • | • | • |

Source: Millennium Ecosystem Assessment

A variety of instruments have been developed to help improve conservation. As noted, the initial approach was a regulatory one, which sought to restrict land uses in particular areas. This approach includes the establishment of protected areas and rules that prohibit farming on sloping land or the use of pesticides in riparian areas. More recently, there have been increasing efforts to use market-based instruments to promote conservation (Landell-Mills and Porras, 2002; Pagiola and others, 2002). These approaches seek to change the behavior of land users by changing their incentives, thus encouraging them to adopt more environmentally benign land uses and discouraging them from adopting more harmful land uses. These approaches include efforts to develop markets for the products of environmentally-

friendly land uses, such as shade-grown coffee; the purchase of easements or direct payments for conservation on private lands; and ‘trading’ systems designed to compensate for damage in one place by improvements elsewhere.

Whatever approach is used, conservation has both costs and benefits. The costs include both the direct costs of implementing conservation measures, and the opportunity costs of foregone uses. The benefits of conservation include preserving the services that ecosystems are providing—although it is important to note that not all conservation approaches conserve all services fully. The question thus immediately arises as to whether the benefits of a given conservation measure justify its costs.

3 Valuing ecosystem services

Economic valuation offers a way to compare the diverse benefits and costs associated with ecosystems, by attempting to measure them and expressing them in a common denominator—typically a monetary unit (see Box 3.1).

Total economic value

Economists typically classify ecosystem goods and services according to how they are used. The main framework used is the Total Economic Value (TEV) approach (Figure 3.1) (Pearce and Warford, 1993). The breakdown and terminology vary from analyst to analyst, but generally include (i) direct use value; (ii) indirect use value; (iii) option value; and (iv) non-use value. The first three are generally referred to together as ‘use value’.¹

- **Direct use values** refer to ecosystem goods and services that are used directly by human beings. They include the value of *consumptive uses* such as harvesting of food products, timber for fuel or construction, and medicinal products and hunting of animals for consumption; and the value of *non-consumptive uses* such as the enjoyment of recreational and cultural activities that do not require harvesting of products. Direct use values are most often enjoyed by people visiting or residing in the ecosystem itself.

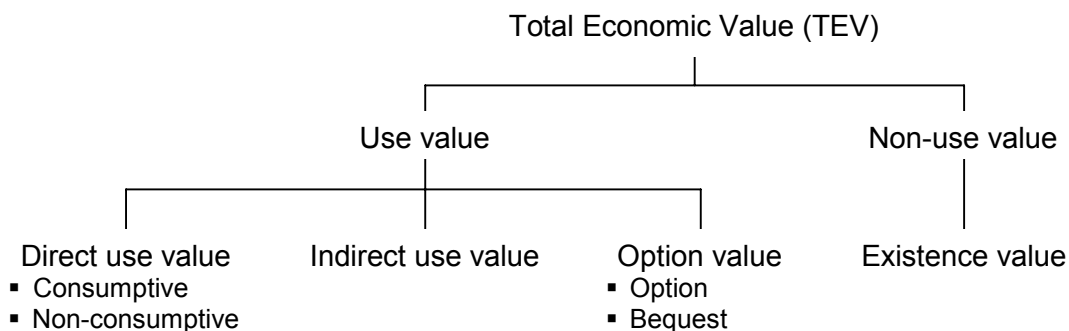
- **Indirect use values** are derived from ecosystem services that provide benefits outside the ecosystem itself. Examples include the natural water filtration function of wetlands, which often benefits people far downstream, the storm protection function of coastal mangrove forests, which benefits coastal properties and infrastructure, and carbon sequestration, which benefits the entire global community by abating climate change. These functions often affect activities that have directly measurable values, allowing their value to be estimated.

- **Option values** are derived from preserving the option to use in the future ecosystem goods and services that may not be used at present, either by oneself (*option value*) or by others/heirs (*bequest value*).² Provisioning, regulating, and cultural services may all form part of option value to the extent that they are not used now but may be used in the future.

- **Non-use values** refer to the enjoyment people may experience simply by knowing that a resource exists even if they never expect to use that resource directly themselves. This kind of value is usually known as *existence value* (or, sometimes, *passive use value*).

Direct use values correspond broadly to the MA’s notion of provisioning and cultural services, while indirect use values correspond broadly to the

Figure 3.1: Typologies of ecosystem services: Total Economic Value



MA's notion of regulating services. Existence value is part of cultural services. The TEV framework does not have any direct analog to the MA's notion of supporting services of ecosystems. Rather, these services are valued indirectly, through their role in enabling the ecosystem to provide provisioning and enriching services.

Box 3.1: Making apples and oranges comparable

Valuation techniques typically express their results in monetary units. This is purely a matter of convenience, in that it uses units that are widely recognized, saves the effort of having to convert values already expressed in monetary terms into some other unit of account, and facilitates comparison with other activities that also contribute to welfare, such as spending on education or health. In particular, use of monetary units expresses the impacts of changes in the services that ecosystems provide in terms of units that are readily understood by decisionmakers and the general public. When all impacts of ecosystem change are expressed in these terms, they can easily be introduced into frameworks such as cost-benefit analysis in order to assess and compare alternative courses of action.

The use of monetary units to compare environmental values emphatically does *not* mean that only services which directly generate monetary benefits are taken into consideration in the valuation process. On the contrary, the essence of practically all work on the valuation of environmental and natural resources has been to find ways to measure benefits which do not enter markets and so have no directly observable monetary benefits.

In general, direct use values are the easiest to value, since they usually involve observable quantities of products whose prices can usually also be observed in the market-place. Recreation is also relatively easy to value as the number of visits is directly observable. Assessing the benefit received by visitors is more difficult, but a large body of literature has developed to tackle this problem, mainly using surveys of tourists' actual travel costs or of their stated willingness to pay to visit particular sites.

Measuring indirect use value is often considerably more difficult than measuring direct use values. For one thing, the 'quantities' of the service being provided—such as the amount of carbon stored in biomass or in the soil—are often hard to measure. While their contribution of ecosystem services to

the production of marketed goods and services may be significant, it is often difficult to distinguish it from that of other, marketed inputs to production. Moreover, many of these services often do not enter markets at all, so that their 'price' is also difficult to establish. The aesthetic benefits provided by a landscape, for example, are non-rival in consumption, meaning that they can be enjoyed by many people without detracting from the enjoyment of others.

Non-use value is the most difficult type of value to estimate, since in most cases it is not, by definition, reflected in people's behavior and is thus almost wholly unobservable (there are some exceptions, such as voluntary contributions that many people make to 'good causes', even when they expect little or no advantage to themselves). Surveys are used to estimate non-use or existence values, such as consumers' stated WTP for the conservation of endangered species or remote ecosystems which they themselves do not use or experience directly.

From an economic perspective, the benefits derived from conserving biological diversity are among the most difficult to define and quantify. While it is relatively easy to identify the benefits obtained from individual components of biodiversity, such as the value of harvesting particular wild species, it is not so easy to describe the benefits of variability itself. Some argue that diverse ecosystems are more resilient and thus provide a kind of natural insurance against climatic and other risks (Perrings, 1998). Others suppose that the likelihood of finding useful products in nature varies with the number of natural expressions considered or, in other words, that diverse ecosystems are more likely to contain economically useful plants, animals or biological compounds (Laird and ten Kate, 2002; Simpson and others, 1994; Barbier and Aylward, 1996; Rausser and Small, 2000). Finally, there is some evidence that the general public including home buyers and tourists prefer variation in ecosystems to homogeneous landscapes (Garrod and Willis, 1992; Powe and others, 1995).

Valuation techniques

Many methods for measuring the utilitarian values of ecosystem services are found in the resource

and environmental economics literature.³ Table 3.1 summarizes the main economic valuation techniques. Some are broadly applicable, some are applicable to specific issues, and some are tailored to particular data sources. A common feature of all methods of economic valuation of ecosystem

services is that they are founded in the theoretical axioms and principles of welfare economics. Most valuation methods measure the demand for a good or service in monetary terms, that is, consumers' willingness to pay (WTP) for a particular benefit, or their willingness to accept (WTA) compensation for

Table 3.1: Main economic valuation techniques

| <i>Methodology</i> | <i>Approach</i> | <i>Applications</i> | <i>Data requirements</i> | <i>Limitations</i> |
|--|--|--|---|---|
| Revealed preference methods | | | | |
| Production function (also known as 'change in productivity') | Trace impact of change in ecosystem services on produced goods | Any impact that affects produced goods | Change in service; impact on production; net value of produced goods | Data on change in service and consequent impact on production often lacking |
| Cost of illness, human capital | Trace impact of change in ecosystem services on morbidity and mortality | Any impact that affects health (e.g. air or water pollution) | Change in service; impact on health (dose-response functions); cost of illness or value of life | Dose-response functions linking environmental conditions to health often lacking; underestimates, as omits preferences for health; value of life cannot be estimated easily |
| Replacement cost (and variants, such as relocation cost) | Use cost of replacing the lost good or service | Any loss of goods or services | Extent of loss of goods or services, cost of replacing them | Tends to over-estimate actual value; should be used with caution |
| Travel cost (TCM) | Derive demand curve from data on actual travel costs | Recreation | Survey to collect monetary and time costs of travel to destination, distance traveled | Limited to recreational benefits; hard to use when trips are to multiple destinations |
| Hedonic pricing | Extract effect of environmental factors on price of goods that include those factors | Air quality, scenic beauty, cultural benefits | Prices and characteristics of goods | Requires vast quantities of data; very sensitive to specification |
| Stated preference methods | | | | |
| Contingent valuation (CV) | Ask respondents directly their WTP for a specified service | Any service | Survey that presents scenario and elicits WTP for specified service | Many potential sources of bias in responses; guidelines exist for reliable application |
| Choice modeling | Ask respondents to choose their preferred option from a set of alternatives with particular attributes | Any service | Survey of respondents | Similar to CV; analysis of the data generated is complex |
| Other methods | | | | |
| Benefits transfer | Use results obtained in one context in a different context | Any for which suitable comparison studies are available | Valuation exercises at another, similar site | Can be very inaccurate, as many factors vary even when contexts seem 'similar'; should be used with caution |

Source: adapted from Pagiola and others, (forthcoming).

its loss (Hanneman, 1991; Shogren and Hayes, 1997).⁴

These valuation techniques have been used extensively in recent years, and a growing literature exists on their application (Hufschmidt and others, 1983; Pearce and Markandya, 1989; Braden and Kolstad, 1991; Pearce, 1993; Dixon and others, 1994; Johansson, 1994; Willis and Corkindale, 1995; Seroa da Motta, 1998, 2001; Garrod and Willis, 1999; Freeman, 2003; Pagiola and others, forthcoming). These techniques can and have been applied to a very wide range of issues (McCracken and Abaza, 2001), including efforts to estimate the benefits of entire ecosystems such as forests (Bishop, 1999; Merlo and Croitoru, forthcoming), wetlands (Barbier and others, 1997; Heimlich and others, 1998; Brander and others,

2003), coral reefs (Cesar, 2000), mangroves (Barbier, 2000), and watersheds (Aylward, 2004; Kaiser and Roumasset, 2002). Other studies have focused on the value of particular ecosystem goods and services such as water (Young and Haveman, 1985), carbon storage (Fankhauser, 1995), non-timber forest products (NTFPs) (Lampietti and Dixon, 1995; Bishop, 1998), recreation (Bockstael and others, 1991; Loomis and Walsh, 1997; Mantua and others, 2001; Herriges and Kling 1999), landscape (Garrod and Willis, 1992; Powe and others, 1995), biodiversity (Pearce and Moran, 1994; Barbier and others, 1995; Pearce and others, 2002), biodiversity for medicinal or industrial uses (Simpson and others, 1994; Barbier and Aylward, 1996; Rausser and Small, 2000), natural crop pollination (Ricketts

Box 3.2: Proceed with caution: Mis-used valuation techniques

All valuation techniques are susceptible to being misused. Two are particularly problematic, however, as they appear to be quite simple to use. Appearances can be deceiving, however. Both of these techniques should only be used with great caution, if at all.

Replacement cost

The replacement cost approach values ecosystem services using the cost of replacing them: either the cost of restoring the ecosystem so that it once again provides the service, or the cost of obtaining the same service in another way. Thus, the water filtration service of a wetland might be valued using the cost of treating water. This technique has been used very widely.

Ever since Shabman and Batie (1978) critiqued early uses of the technique, it has been generally accepted that its validity requires three conditions to hold: (i) that the replacement service be equivalent in quality and magnitude to the ecosystem service; (ii) that the replacement be the least cost way of replacing the service; and (iii) that people would actually be willing to pay the replacement cost to obtain the service.

In practice, however, few have verified whether these conditions hold. Without doing so, it is very easy to over-estimate the value of ecosystem services, perhaps by a large amount.

For further discussion, see Freeman (2003).

Benefits transfer

'Benefits transfer' refers to the use of valuation estimates obtained (by whatever method) in one context to estimate values in a different context. For example, an estimate of the benefit obtained by tourists viewing

wildlife in one park might be used to estimate the benefit obtained from viewing wildlife in a different park. Alternatively, the relationship used to estimate the benefits in one case might be applied in another, in conjunction with some data from the site of interest ('benefit function transfer'). For example, a relationship that estimates tourism benefits in one park, based in part on tourist attributes such as income or national origin, could be applied in another park, using data on income and national origin of the latter park's visitors.

Benefits transfer is a seductive approach, as it is cheap and fast. It has been the subject of considerable controversy in the economics literature, however, as it has often been used inappropriately. Case Study 4 illustrates how dangerous it can be: even within a narrowly-defined environment (forests in coastal Croatia, an area of about 5,000 km²), the benefits of ecosystem services can differ by an order of magnitude. A consensus seems to be emerging that benefit transfer can provide valid and reliable estimates under certain conditions. These include the requirement that the commodity or service being valued should be very similar at the site where the original estimates were made and the site where they are applied; and that the populations affected should also have very similar characteristics. Of course, the original estimates must themselves be reliable for any attempt at transfer to be meaningful.

For further discussion see, among others, Brookshire and Neill (1992), Brouwer and Spaninks (1999), and Barton and Mourato (2003).

and others, 2004), and cultural benefits (Pagiola, 1996; Navrud and Ready, 2002). Many valuation studies are cataloged in the Environmental Valuation Reference Inventory (EVRI) website, maintained by Environment Canada (EVRI, 2004).

Some of these techniques are widely used, others only selectively so (Dixon and others, 1994). Some techniques, such as benefits transfer and replacement cost have tended to be used incorrectly and inappropriately, to the point that many economists advise against using them in all but exceptional circumstances (Box 3.2).

It is important to use these valuation techniques properly. They provide powerful tools to assess the value of particular ecosystem benefits, but if they are mis-applied, their results will be of little use. This paper does not provide detailed guidance on using these techniques; many other sources do so. Rather, the purpose of this paper is to help decisionmakers frame the valuation question properly to ensure that the numbers these techniques

provide are relevant and useful for addressing specific policy issues.

Notes

- ¹ Option value is sometimes grouped with existence value as a kind of non-use value, but the interpretation is not otherwise different.
- ² Some analysts add *quasi-option* value: the value of avoiding irreversible decisions until new information reveals whether certain ecosystem services have values we are not currently aware of (Arrow and Fisher, 1974).
- ³ Hanneman (1992) and Pearce (2002) review the history of economic valuation of environmental impacts.
- ⁴ Broadly speaking, WTP is appropriate when beneficiaries do not own the resource providing the service or when service levels are being increased, while WTA is appropriate when beneficiaries own the resource providing the service or when service levels are being reduced. In practice, WTA estimates tend to be substantially higher than WTP estimates (Hanneman, 1991).

4 Valuing the total flow of benefits

Policymaking requires an accurate assessment of the state of the national economy at any point in time. Unfortunately, the information available is seriously incomplete. Despite the importance of ecosystem services to the economy, their contribution is hard to discern in the available statistics. The first approach to ecosystem valuation aims at clarifying the contributions that ecosystems make to economic activity. This approach is most appropriately used to answer questions such as: What benefits do protected areas provide to society? What is the contribution of ecosystem services to the national or a local economy and to the welfare of people living there? What are the benefits of specific ecosystems, such as forests?

Why are we doing this?

Policymakers receive a large number of indicators on the economic benefits generated by various sectors of the economy. The single most important indicator of an economy's size is Gross Domestic Product (GDP), that is, the total market value of all goods and services produced within the political boundaries of an economy during a given period of time, usually one year.¹ Some of the components of GDP can be interpreted as measuring certain ecosystem benefits (for example, the output of natural resource based activities such as logging or fishing). But these indicators are problematic on several grounds:

- they are *incomplete* in that many ecosystem benefits go completely un-measured (direct use values are most likely to be included, but only to the extent that they enter markets; informal collection of NTFPs, for example, is seldom reflected in national accounts);

- they often *mis-attribute* ecosystem benefits to other sectors (this is particularly true of indirect use values: the water regulation benefits of wetlands, for example, do not appear as benefits of wetlands but as higher profits in water-using sectors); and
- they are often *misleading* in that certain benefits may appear exaggeratedly high (for example, if fishing rates are far in excess of natural growth) while others do not appear at all.

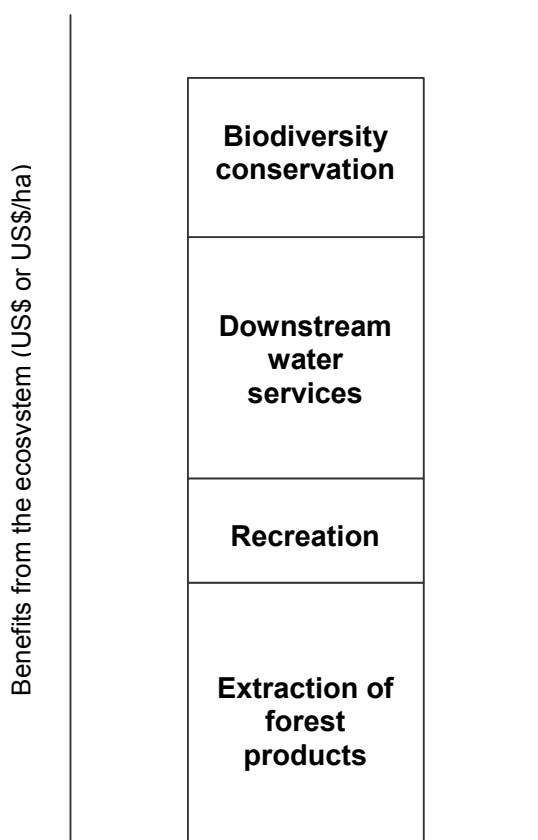
As a result, existing indicators tend to vastly undervalue the benefits provided by natural ecosystems. Economic valuation can help to develop better indicators that provide a more accurate picture of how ecosystems, individually and collectively, contribute to the economic welfare of society.

It is helpful to begin by understanding why we care about measuring 'product' or 'national income'. The generation of goods and services, that is the 'product', in an economy serves two purposes: current consumption, which directly contributes to our welfare (economists call this 'utility'), and investment for future production and consumption, which can contribute to our future welfare. Mis-measuring ecosystem benefits means that we are mis-measuring the resources available to us. Their loss would likewise tend to go un-noticed. This would be a problem even if the contribution were relatively minor; but in the case of ecosystem services, the contribution is often substantial, making their omission from common indicators extremely harmful. This omission can easily lead to mis-management. There will be little concern about preserving ecosystems that appear to contribute little to economic activity.

The first approach to valuing an ecosystem, then, is to value the total flow of benefits that it is providing. This approach attempts to estimate the total net benefit that ecosystems are providing, typically on an annual basis. It can be applied at a number of scales, from that of a single patch of ecosystem in a specified location, to that of entire ecosystems, or groupings of ecosystems, to conceivably all ecosystems on the planet (although, as will be discussed, results are increasingly problematic both empirically and conceptually as the scale increases). Efforts to estimate the value of natural capital are a variant of this approach.

If the value of the various benefits could be estimated, the result would look something like Figure 4.1, which shows four examples of services for the sake of illustration (forest products are an example of a consumptive direct use, recreation is a non-consumptive direct use, watershed protection is an indirect use, and biodiversity conservation is often considered to provide existence value, although aspects of its could also

Figure 4.1: Flow of benefits from an ecosystem



be considered as an indirect use or as option value). The results could be presented either in terms of total benefits of that ecosystem, or in terms of per hectare benefits. An estimate in total terms would be better suited to a comparison to GDP numbers, while an estimate expressed in per hectare terms would be better suited to comparison to alternative land uses.

How do we do it?

There have been very few efforts to date to actually measure the total flow of benefits of entire ecosystems. The TEV framework has been much used as a heuristic device, but there have been few efforts to actually estimate the TEV of particular ecosystems. Two of the main such efforts involve forests, in Mexico (Adger and others, 1995) and in the Mediterranean (Merlo and Croitoru, forthcoming). Most studies have focused on a single service provided by a given ecosystem. Some of these look at current flows (or flows in a specific base year), while others seek to predict current and future flows. These studies have usually been conducted on a one-time basis. These studies give only part of the picture, but provide building blocks which might be used to construct a more comprehensive view of the ecosystem—although as noted below, these are building blocks that often do not fit well together.

Identifying benefits

As the objective is to assess the flow of all benefits provided by an ecosystem, the first step is to identify the specific services it provides. There are several possible typologies that might be used as the basis for this (see Chapters 2 and 3).

In identifying benefits, it is important to bear in mind that many ecosystems can have impacts at some distance from where they are located. Upstream watersheds can affect human populations (and other ecosystems) far downstream, for example, and mangrove forests and coral reefs serve as breeding grounds and nurseries for fish species that range widely. The links between ecosystems and their effects on welfare can also involve substantial lags. Erosion in an upper watershed may only reach downstream reservoirs and other vulnerable infrastructure after several years.

Drawing the spatial and temporal links boundaries appropriately is a critical, but difficult step.

The typical approach to doing this kind of valuation is to seek studies that have focused on each of the individual services that an ecosystem provides, and then adding up the relevant results. This approach faces many pitfalls:

- Available studies are often very un-systematic, with some benefits being studied extensively and others very little. Extractive uses often have the best data, although this is not necessarily true of many NTFPs. Recreation has also been studied quite exhaustively. Indirect use benefits, on the other hand, have only occasionally been studied, and option and existence values even less. This is illustrated in Case Study 1, for example: while the direct use values of Mediterranean forests could be estimated in all countries, only some indirect use values could be estimated.
- Available studies may be un-representative. Studies focusing on a particular benefit often seek out sites where that benefit is prominent. Extrapolating these results to the entire ecosystem can be very misleading.
- Available studies may ignore other benefits and their effect on each other. Most studies make a concerted effort to hold other effects constant. This may be good science in the narrow context of the study, but it complicates efforts to use the results in the real world, where everything else most assuredly is not constant. The results of a study that examines water benefits assuming no logging cannot be added to the results of a study of the benefits of logging. These two activities may well be mutually inconsistent.
- Available studies often use incompatible units. In assessing the benefits of a given ecosystem, a per hectare value is usually required. But recreation studies usually report benefits per visitor, and water use studies report benefits per cubic meter of water. Some of these conversions are relatively simple—in the case of a protected area of known size and with known visitor numbers, for example, converting from per visitor benefits to per hectare benefits may be straightforward, although even here many questions arise: does one include the buffer zones? What

about the biological corridors that link the protected areas together? And should core zones be weighted more than mixed-use or buffer zones? In most cases, however, conversion is far from trivial.

Distributional issues, to be addressed more fully in Chapter 6, already appear: if the analysis is undertaken from the perspective of the country where the ecosystem is located, then the value of certain services whose benefits are primarily outside the country should not be counted. Thus, services such as biodiversity conservation and carbon sequestration should usually not be counted unless the country receives payments for providing them. In Case Study 8, for example, biodiversity conservation benefits in Madagascar are valued using payments received from the international community, and carbon sequestration benefits are not included as the country is not currently receiving any payments for this service. Some direct uses should not be counted, either: in particular, welfare benefits to foreign visitors to a national park should not be counted, except to the extent that they are captured in country through fees or other devices. The case of Madagascar illustrates this as well: the tourism benefits to the country are not valued using the consumer surplus of international visitors, but only the fees that they pay.² Conversely, if the analysis is undertaken from the perspective of humanity as a whole, then all these benefits should be counted.

Quantities

In general, valuation of each benefit involves first estimating the quantities of the good or service being provided (for example, the amount of wood being harvested, the supply of clean water to downstream users) and then multiplying that by an estimate of its value and subtracting any costs involved in using the service (for example, the harvesting costs for timber or NTFPs). Although the procedure sounds simple, there are numerous issues to consider.

Actual vs. potential value. In many cases, only a small portion of the available flow of services is used. Thus in any given case, only a fraction of available goods might in fact be harvested, and only a fraction of available water flow might be

Box 4.1: How much are pine kernels worth?

Valuation efforts must always be subjected to a sanity check: do the results make sense? The case of pine kernels in Lebanon illustrates some of the potential problems. Pine kernels are a high-yield, high-value NTFP. Multiplying available estimates of yield (480 kg/ha), price (US\$20.3/kg), and area (5,400ha) results in a total value of about US\$52 million. This is equivalent to about US\$9,600/ha of stone pine area, or US\$390/ha if spread over Lebanon's entire forest area. If correct, this result would be truly astounding: pine nuts in Lebanon by themselves would have a greater value, on a per hectare basis, than the entire estimated TEV of forests in any other Mediterranean country (see Case Study 1). Even considering that this is a gross rather than a net value, there is good reason to suspect that something is amiss. Perhaps the available yield data reflect optimal conditions, and are

Source: Croitoru and Merlo (forthcoming)

not representative; perhaps the available price data include costs other than raw materials, or apply to high-quality products that only represent a small share of total output; or perhaps the quantity measure used in the yield data does not correspond to that used in the price data, because of processing losses or wastage along the production chain. All of these could lead to over-estimation. It may also be that the individual data are correct, but that only a small part of total production is marketed. In that case, prices would probably plunge if all potential production was brought to market.

Seemingly anomalous results such as these are not necessarily wrong, but they need to be checked carefully; exceptional results should be subjected to higher standards of proof. Unfortunately, problems with valuation are not always so easy to spot.

used. Should the same value be assigned to products that are not, in fact, used? The issue arises when we have an estimate of the average benefits from a resource and we need to apply it to a particular site. The example of NTFPs illustrates this. In a well-known study, Peters and others (1989) estimated the value of a hectare of forest in the Peruvian Amazon as being almost US\$700/ha, based on the products that could potentially be harvested from it (including timber, rubber, fruits, and nuts). But only a small fraction of this potential production is actually harvested. Harvesting more would likely cause prices to plunge as supply increases.

Gross vs net value. Many studies fail to consider the cost of using services. A fruit hanging on the branch of a tree only becomes a valuable NTFP once it is harvested and brought to market. Doing so is not costless. Failure to consider these costs can result in a very substantial over-estimate of the potential value of the service. It is also important to incorporate these costs realistically. The study by Peters and others (1989), for example, assumed that harvest costs for NTFPs were a percentage of revenue. Under this assumption, harvest will clearly always be profitable. Yet it is not hard to imagine that, high transport costs would make extraction unprofitable in much of the Amazon.

Current flows may not be sustainable. For example, the level of extraction of timber and NTFPs may be higher than the rate of growth of these products, thus diminishing future capacity to produce them. Formally, one should subtract from the current benefit flow a 'user cost'—the reduced future benefit (in the form of higher harvest costs or reduced harvest revenue) resulting from over-extraction today. At the very least, the fact that current benefit flows may not be sustainable should be noted). An alternative approach would be to estimate what the current benefit flow would be under the restriction that only sustainable uses are allowed.

Box 4.1 describes one case in which what appears to be a straightforward price times quantity calculation yields results that are unlikely to be correct. Results should always be tested for reasonableness. Trumpeting unrealistically high estimates can easily undermine the credibility of the results.

Prices

On the price side, many goods and services have observable prices, and these can be used in the calculations. Even when prices cannot be observed (for example, products harvested for home consumption), there are generally-accepted and reliable ways to estimate the value of the products (for example, by using the value of close sub-

Box 4.2: Of diamonds and water

There is a well-known paradox in economics called the ‘diamonds and water paradox’: Water, despite its importance for survival itself, is generally very cheap, while diamonds, despite their relative unimportance except as an adornment, tend to be very expensive. The reason for this paradox lies in the relative abundance of water and diamonds. Water is generally plentiful, and so an additional unit tends to be cheap. Diamonds, on the other hand, are scarce, and so command a high price.

How is this relevant to the valuation of ecosystem services? Most valuation studies of services such as water supply have been undertaken in contexts where these services are relatively abundant. Even in cases

stitutes). For other services, approaches such as a CV can be used to elicit value information directly from users.

An important caveat to the use of observed prices is that they may well be distorted by policy interventions or other problems. In that case, they will not properly reflect how resources contribute to society. This will, in turn, distort the valuation of ecosystem services. For example, if irrigated agriculture is highly subsidized, the value of watershed protection services will appear higher than it really is. When the analysis is undertaken from the perspective of society, therefore, it is important to adjust observed markets to correct for any distortions. We return to this issue in Chapter 6.

All of these approaches face increasing problems, however, as the scale of the analysis increases. The thought experiment that underlies this approach to valuation is ‘how much worse off would we be if we did not have the ecosystem, and all the services it provides, at all?’ This question is reasonably well defined at small scales (a particular forest, say, in a specific country): the total physical flow of goods such as timber is small enough that its presence or absence is unlikely to affect prices. As the scale of the analysis increases (to cover, for example, all tropical forests, or an entire country), this assumption is less and less likely to hold, and observed prices are less and less likely to be applicable (see Box 4.2). This is one of the reasons that efforts to value all the world’s ecosystems can produce non-sensical results (see Box 4.3).

where water is considered scarce (usually defined as availability of less than 1,000m³/person/year) it is not usually so scarce as to endanger life itself. Using these results to estimate the value of services provided by small-scale ecosystems is appropriate. But as we start considering the value of all services provided by ecosystems on a large scale, the premises of the paradox no longer hold. If we had no water at all, it would be extraordinarily valuable. So if we consider the value of all water provided by a large ecosystem, or all freshwater on the planet, the marginal price for an extra unit of water is no longer a reliable guide.

How do we use the results?

Such studies provide two important insights. First, they can demonstrate that seemingly ‘worthless’ land uses may in fact be quite important to the economy. They can clarify the relative importance of ecosystem services to total economic output, and thus guide overall investment strategy, although more detailed analysis is usually required to assess specific interventions, as discussed below.

Second, the composition of benefits provides an indication of how likely it is that ecosystems are being managed optimally. Land use decisions are generally made by groups who mainly receive direct use benefits. Such groups often have strong incentives to manage land so as to maximize direct use benefits, and pay little or no attention to the consequences for other benefits. Thus, the greater the share of ecosystems benefits provided by indirect, option, or existence value, the less likely it is that the ecosystem is being used optimally.

It is important to note that the estimates are specific to a given management—usually current management practices, if the estimates are based on observed data. If management were to change, the benefit flows would change, and thus so would the value of the ecosystem.

Care is needed in interpreting current flows of benefits from an ecosystem. If the ecosystem is not being managed sustainably, these flows may well decline in the future. High rates of extraction of products such as timber, for example, may not

Box 4.3: “A serious underestimate of infinity”

In a landmark paper published in *Nature* in 1997, Costanza and others (1997) attempted to calculate the total value of all ecosystems on earth. Using a range of estimates of the value of individual ecosystems and scaling them up according to the total area covered by each such ecosystem globally, they arrived at an estimate of the total value of all ecosystem services ranging from US\$16-54 trillion a year, with a central estimate of US\$33 trillion (in 1997 prices).

This paper has had a significant impact and its results have been widely quoted by scientists and environmentalists. However, most economists consider it to be profoundly flawed, both conceptually and methodologically.

- The study generates its global estimates by extrapolating the results of valuation studies undertaken in specific locations. As discussed in Box 3.2 and illustrated in Case Study 4, such ‘benefits transfers’ are often unreliable due to wide variation in ecosystem values across different sites.
- The study uses estimates of average value based on marginal changes in ecosystem services to calculate the aggregate value of entire ecosystems. However, this approach fails to account for the variation in unit values as the scale of analysis changes. As discussed in Box 4.2, the value ascribed to a resource depends on whether the change in availability being contemplated is large or small.
- The study results exceed the sum total of global economic income recorded in 1997. They cannot be interpreted as an estimate of society’s willingness to pay, as one cannot plausibly pay more than the total value of all income. The results also cannot be interpreted as society’s willingness to accept compensation for loss of all the world’s ecosystem services. Without these services, we would all be dead, so there is no finite compensation we would accept for the loss of all ecosystem services. In this sense, the results are “a serious underestimate of infinity,” in the words of Michael Toman.
- The study suggests that its results be used to guide policy decisions. But, as discussed throughout this paper, information on total benefit flows, even if accurate, cannot provide guidance on specific conservation decisions, which are about making incremental changes in those flows.

Sources: Costanza and others (1997), Toman (1998), Bockstael and others (2000).

be sustainable if extraction exceeds the natural rate of growth. High apparent flows of current

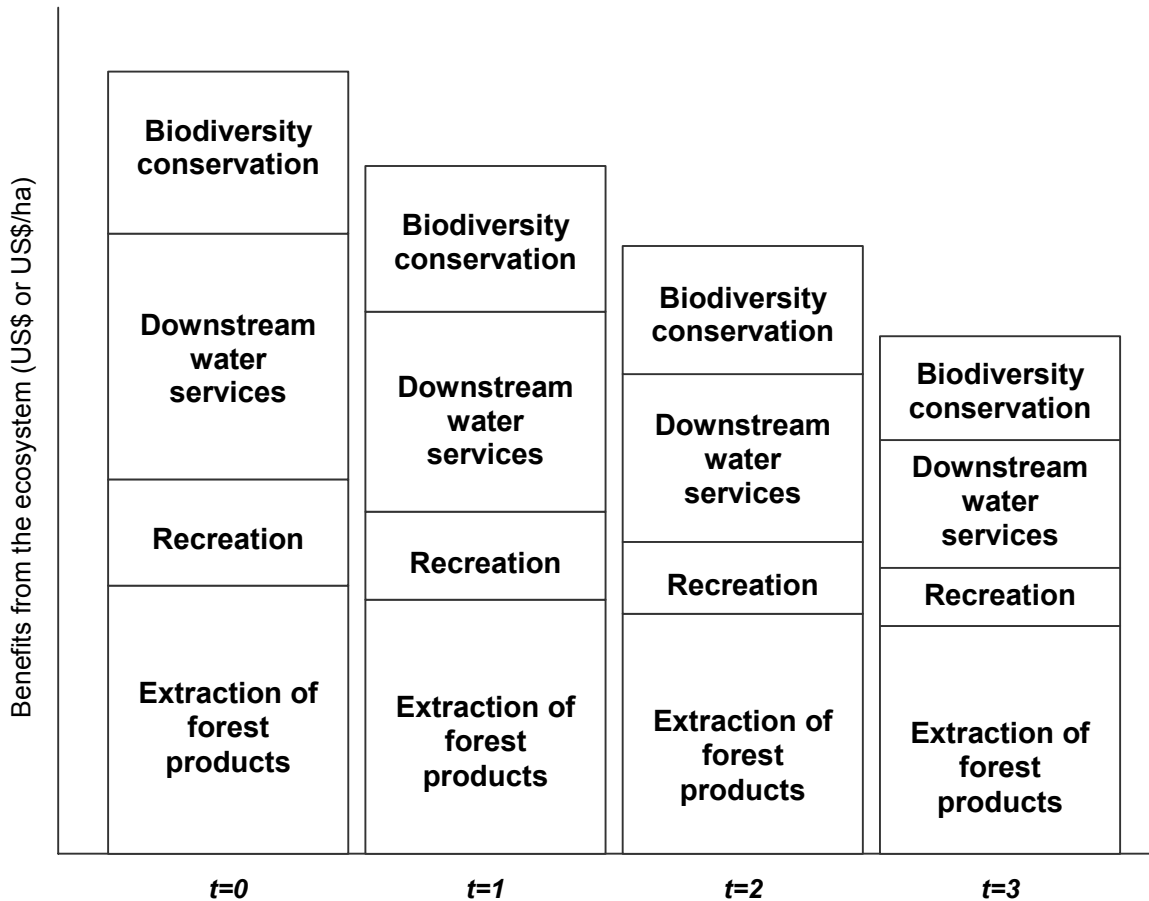
benefits may thus come at the expense of future flows. (Conversely, if extraction is less than natural growth, the stock of the resource would grow over time; in this case the current flow of benefits would tend to understate potential future benefits.) Interpretation is also difficult when examining the value of ecosystems on a large scale, as discussed in Box 4.2.

Natural capital

Efforts to value ‘natural capital’ are a variation of this approach. A nation’s wealth has traditionally been measured as the sum of produced capital, that is, machinery, equipment, and infrastructure (such as buildings, roads, and ports) and commercial land. But ecosystems can also be considered a form of capital. Forests represent wealth in terms of the flow of timber and non-timber products and services they provide. Fish stocks provide consumption benefits. Just as the stock of produced capital determines how much industrial production a country can undertake, so will the stock of natural capital determine how many ecosystem services it will receive. Ecosystems, considered as a form of natural capital, have the advantage that, unlike produced capital, they can regenerate themselves—if they are managed appropriately. But like produced capital, natural capital is subject to depletion which reduces future production possibilities. In the case of forests, for example, harvest rates that are greater than the rate of growth will come at the expense of the stock of the resource. This will undermine future harvests, as well as any other services that depend on the extent of forests in the ecosystem. Likewise, overharvesting fish can lead to a collapse of the fishery, as has already happened several times (Jackson and others, 2001).

Estimates of the value of ecosystems as natural capital are very closely related to the estimates of the flow of benefits they provide. Rather than looking at the flow of benefits from an ecosystem in a single year, the natural capital approach considers the present value of all current and future benefits that the ecosystem will generate. Estimating this value requires projecting how the flow of services, and their value, would evolve over time. This is illustrated in Figure 4.2, for a case in which an ecosystem is degrading and the services it provides are gradually diminishing.

Figure 4.2: Estimating the value of an ecosystem as natural capital



A major contribution of wealth accounting is that it highlights the main sources of income. Income generation in the most advanced countries heavily depends on human capital (the level of skills and education of the population). Developing countries depend more heavily on natural resources. Case Study 2 shows the results of a study of the wealth of Sub-Saharan Africa. Natural capital is an important part of this wealth, and ecosystems provide the majority of this (World Bank, 2004). Wealth accounting points at areas in which careful management of resources is required. For example, attention to land degradation in Africa appears as a very high priority, if welfare is to be sustained over time.

Beyond emphasizing the different sources of wealth and the attendant priorities for their management, wealth accounting provides important indicators of the sustainability of development. It does this first by clarifying what is truly income—depletion of minerals, reductions in fish stocks or

forest resources, for example, represent asset liquidation rather than income. This distinction can be important in judging the economic performance of the most resource-dependent economies. Most important, the adjusted net saving (ANS)³ of a country indicates whether social welfare is rising or falling. ANS measures the change in total wealth in a given period; it is calculated by adjusting the traditional measure of net national savings to account for activities which enhance wealth, such as education expenditure (an investment in human capital), as well as activities that reduce wealth, such as depletion of mineral and energy reserves, depletion of forests, and damages from pollution. A negative ANS indicates that an economy is on an unsustainable path. Case Study 3 shows how adjusting for depletion of forests can affect the estimated savings rate in Ghana.

In addition to providing an important tool for the analysis of sustainability, asset accounts can also indicate the sorts of policy interventions which

may be needed in order to place development onto a more sustainable path. Boosting saving rates, for example, can certainly be achieved by the monetary and fiscal policies of ministries of Finance. But asset accounting also suggests that the fundamental rate of saving can also be boosted by better resource management policies, in particular policies which discourage excessive rates of exploitation and damage. Similarly, in over-polluted countries, policies aimed at bringing down pollution levels to the point where marginal costs and benefits are being equalized will have the effect of boosting measured genuine saving, an indication that the economy is on a more sustainable path.

Notes

- ¹ Gross National Income (GNI), another commonly used indicator, adds net receipts of primary income from abroad. GNI was formerly known as Gross National Product (GNP).
- ² The country receives additional benefits from the economic activity that the tourists stimulate through their demand for services. Tourist spending is often used as a measure of this benefit. It is not. The country needs to devote substantial amounts of resources to supply tourists with lodging, food, transport, and other resources. Tourist spending ignores these costs, thus vastly over-estimating the contributions tourists make.
- ³ Sometimes known as 'genuine' savings (Hamilton and Clemens, 1999).

5 Valuing changes in flows

Estimates of the total annual flow of benefits from an ecosystem have frequently been used to justify spending to address threats or to improve its condition. But using such value estimates in this way would be a mistake. To examine the consequences of ecosystem degradation, or to assess the benefits of a conservation intervention, it is not enough to know the total flow of benefits. Rather, what is needed is information on how that flow of benefits would *change*.

The second approach to valuing ecosystem speaks directly to policy concerns: this approach attempts to estimate the *change* in the total net benefit that ecosystems would provide as a result of an intervention.¹ This approach can be applied to assess the likely results of a deliberate intervention, or to examine the consequences of on-going trends such as deforestation. The scale of the analysis is determined by the scale of the intervention being considered.

Why are we doing this?

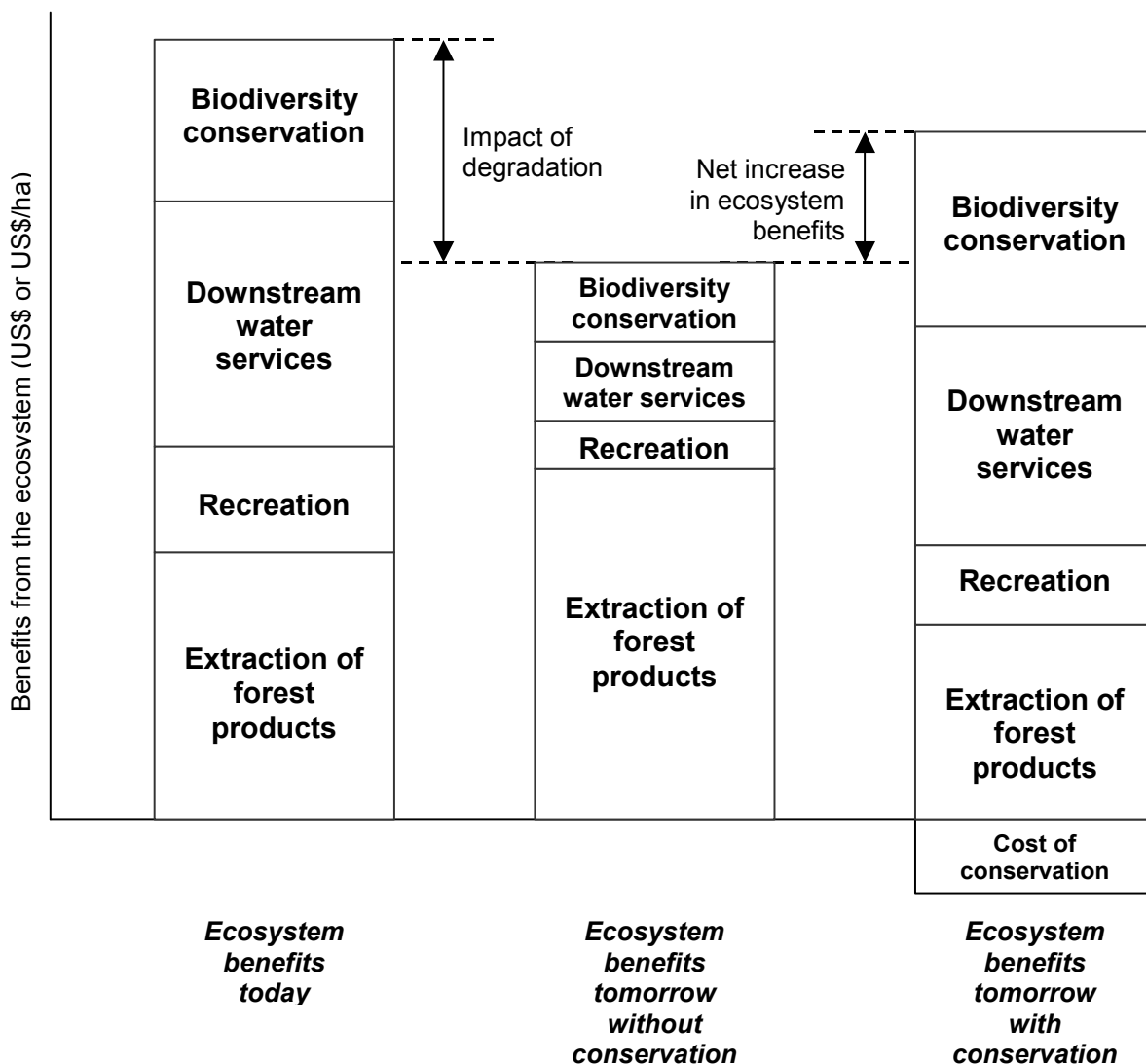
Measures of the current flows of benefits provided by ecosystems provide useful and interesting information on how things stand, but they are not generally directly policy-relevant. It is a common mistake to conclude from the fact that an ecosystem is currently providing valuable services that it is worth spending a lot of resources to conserve it. To assess whether a specific conservation intervention is worth undertaking, we must know two things: what would happen if we did nothing? And what would happen if we did intervene in a specific way? By using the entire flow of benefits as a yardstick for policy decisions, we are implicitly assuming that doing nothing would result in the complete and instantaneous loss of all ecosystem services, and that conversely conservation would result in the complete and instantaneous halt of all degradation processes. Neither assumption is realistic.

The consequences of inaction are not necessarily severe. Some ecosystems may not be threatened by degradation; they might, for example, be too remote, or their soils might be too poor to make them attractive for agriculture. Others may be subject to forms of degradation that do not significantly affect their main benefits. Excessive hunting, for example, might depopulate a forest and render it much poorer from a wildlife perspective, but leave watershed protection services largely intact.

Even when degradation is a real problem, it is rare for all ecosystem services to be lost entirely. A forested watershed that is logged and converted to agriculture, for example, will still provide a mix of environmental services, even though both the mix and the magnitude of specific services will have changed. It would be a mistake, therefore, to credit a conservation project which prevents such degradation with the *total* value of the flow of benefits provided by the ecosystem at risk. Rather, what is needed is an assessment of the incremental change in the value of services provided by the ecosystem resulting from a well-defined change in how it is managed. Where the change does involve the complete elimination of ecosystem services, such as the conversion of an ecosystem through urban expansion, then the change in value would equal the total economic value of the services provided by the ecosystem. This is the exception, however, rather than the rule.

Finally, ‘degradation’ might involve conversion to another use which, although it results in the loss of some benefits, provides other benefits in exchange. Converting forests to agriculture results in loss of biodiversity, in reduced watershed protection (if done badly), and in the loss of recreational benefits, but in exchange it will provide increased food production, which will at least

Figure 5.1: Change in ecosystem benefits resulting from a conservation project



partly compensate for the loss of the other services.

Likewise, conservation interventions are not necessarily fully effective. Some interventions may only succeed in slowing rather than halting degradation. This does not necessarily mean that they are not worth undertaking, however; they may be, if their benefits are sufficiently high. Conversely, some interventions may not only halt degradation but actually improve conditions. And just as degradation can affect the various categories of services that an ecosystem provides differently, so can conservation. Some services may improve even as others are reduced. What is needed, then, is an assessment of the net impact of

the various changes. Finally, the cost of implementing the conservation measures themselves must be taken into account

The question is whether the total economic value of the services provided by an ecosystem managed in one way (with conservation) is more or less than the total value generated by the ecosystem if it were managed in another way (without conservation), after allowing for the cost of changing management (implementing the conservation measures). It is quite likely that a change in management will increase the value of some services and decrease the value of others; what matters is the net difference between the total value of all services.

Figure 5.2: Cost-benefit analysis of a conservation project

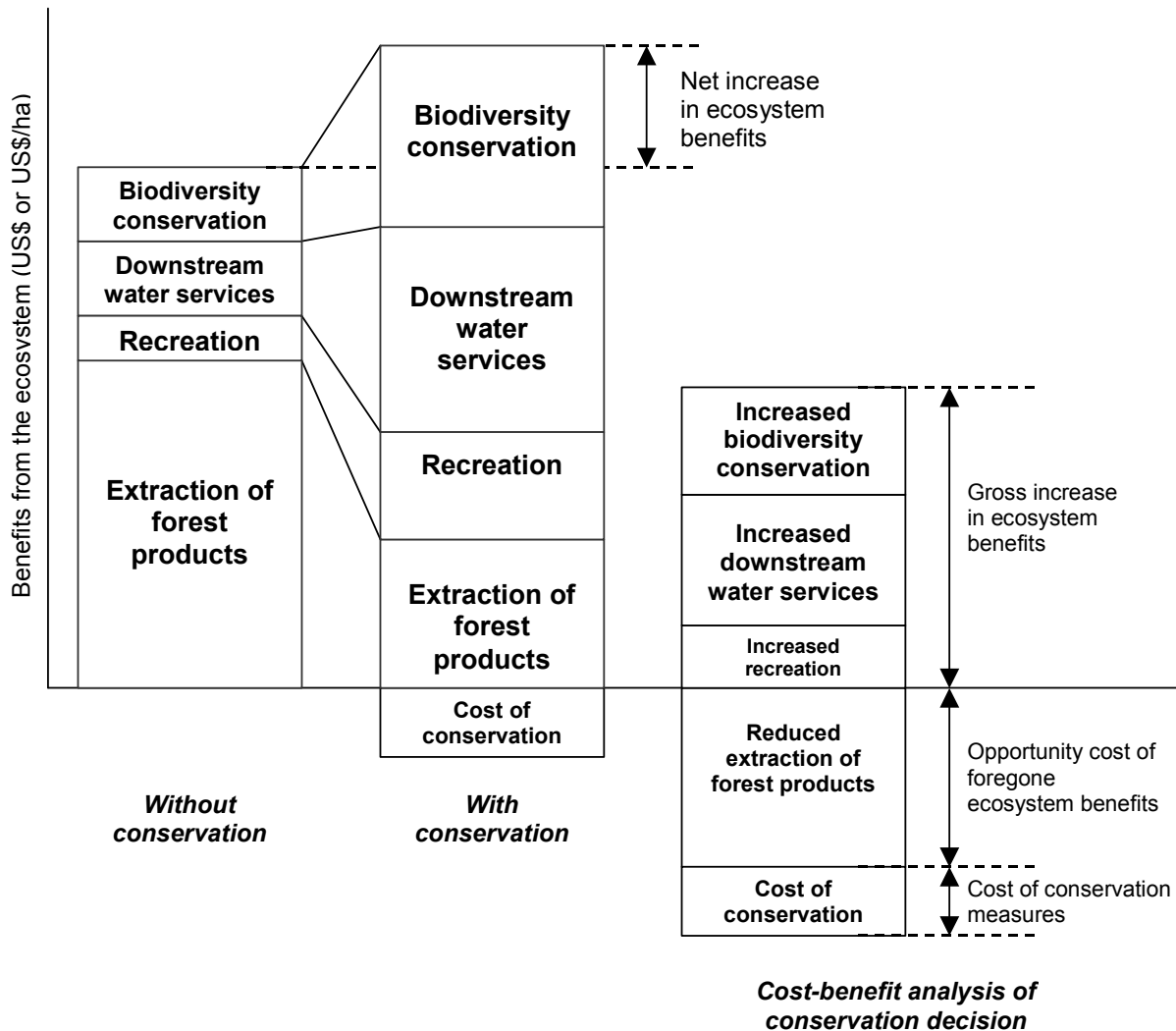


Figure 5.1 illustrates this approach. The first column on the left shows the value of the total flows of benefits that an ecosystem is providing today, as might have been estimated in the previous chapter. The pattern of use is assumed to be unsustainable: the ecosystem is being degraded, perhaps by excessive extraction of forest products, reducing its future capacity to provide services. (This problem is not, of course, visible simply by looking at the results—one of the limitations of estimates of total current flows.) The next two columns show two alternative tomorrows. The middle column illustrates what might happen under current degradation trends, if nothing is done. The overall value of services provided by the ecosystem declines. Note that the decline is not uniform across all services; indeed, the value

of some services may actually increase. The case illustrated here is one in which the high level of extraction of forest products is the reason for the overall degradation. As extraction rises, recreational opportunities, watershed services, and biodiversity conservation are all diminished. Those who extract forest products may be better off, but society as a whole is worse off. The difference between this column and the previous column can be considered to be the impact of degradation, and valuation can put a figure on this impact.² The third column shows the value of the services the ecosystem would provide if it were conserved. Note here that conservation in this example is not fully effective: the overall value of services still falls compared to the present situation. However, conservation does succeed in improving condi-

tions over what they would have been otherwise, although it does this at a cost. As illustrated, the conservation measures (whose cost is shown as a negative value) severely restrict the extraction of forest products. By doing so, they preserve a good part of the recreational, watershed protection, and biodiversity conservation services the ecosystem is providing. The difference between this column and the 'without conservation' column can be taken as the benefit of conservation. A cost-benefit analysis of whether to undertake the conservation measures would compare this value to the cost of undertaking them.

The value of conservation is not necessarily positive. There may be cases in which the value of the additional services obtained by converting an ecosystem to an alternative use exceeds the value of the services obtained under conservation. The change in value must then be compared to its cost in order to determine whether it is worth undertaking, from an economic perspective. This, too, may yield a negative result: conservation is not always the preferred option, from an economic perspective.

A critical point illustrated in Figure 5.1 is that this analysis should not compare ecosystem benefits before and after conservation measures are implemented, as many other factors may also have changed in the intervening period. Rather, it should compare ecosystem benefits with and without the conservation measures: that is, it must compare what would happen if conservation measures are implemented to what would happen if they are not.

Figure 5.2 illustrates this same approach in a different way. Here the values of the various services that would be obtained with and without conservation are compared directly. Some services are increased thanks to conservation, while others (in this case, extraction of forest products) are reduced. The third column shows the net changes in each service, along with the cost of conservation. This presentation illustrates the fact that the cost of conservation actually has two components: the direct, out-of-pocket costs of implementing the conservation measures themselves, and the opportunity cost of the foregone benefits from the services whose use is restricted. These two costs should then be compared to the

gross increase in ecosystem benefits that would result from implementing the conservation measures. It is a very common mistake to consider only the out-of-pocket costs of conservation, ignoring the opportunity costs.

Case Study 4 shows the result of one analysis presented in this format. The results show that both costs and benefits vary by site, depending on local characteristics. Sites on steeper slopes, for example, are costlier to reforest. Thus even within the same county, benefits can vary by several orders of magnitude: the average benefit of US\$790/ha (discounted at 10 percent) masks substantial variation in the net benefits of reforestation. As can be seen, some benefits were not found at some of the sites. Some of the sites had no expected erosion protection benefits, not because reforestation wouldn't reduce erosion, but because there were no downstream facilities at risk from erosion. At those sites, therefore, erosion reduction benefits could be ignored, as they were negligible.

How do we do this?

Such an assessment can be undertaken either by explicitly estimating the change in value arising from a change in management, or by separately estimating the value of ecosystem services under the current and alternative management regimes, and then comparing them. If the loss of a particular ecosystem service is irreversible, then the loss of the option value of that service should also be included in the analysis.

Estimating changes in ecosystem benefits and costs is sometimes easier than estimating the value of the total flow of benefits of an ecosystem, because the analysis can focus on only those benefits and costs which are affected by the proposed conservation action. In many cases, available data or expert opinion can strongly indicate that certain types of benefits are unlikely to be affected by the proposed action. The scope of the required study is then considerably narrowed to just those benefits that are expected to be affected by the change.

The key challenge is to identify the changes that would result from the proposed action. How much more production would there be? To what extent would downstream water supplies be improved?

As already discussed, implicitly assuming that *all* benefits would be lost in the absence of conservation, or conversely that *all* benefits are due to the conservation action are common mistakes.

For those services which are expected to be affected, the main challenge usually involves estimating changes in quantities. How much cleaner will water be if a watershed is reforested? This requires knowing how water quality is related to vegetation cover in the watershed. This is by no means trivial. The linkages between land use and water services, for example, are much less understood than is commonly supposed (see Box 5.1). Several methodologies have been developed to assess changes in carbon sequestration (UNECE-FAO, 2000; IPCC, 2001).

Box 5.1: Quantifying changes in water services

Ecosystems such as forests are widely believed to provide a variety of hydrological services, including reducing erosion, thus reducing sediment loads in waterways; regulating the timing of waterflows, thus reducing flood risk and dry season water shortages; increasing the volume of available water; and improving water quality. The evidence on these links is often far from clear, however (Bruijnzeel, 1990, 2002, 2004; Calder, 1999; Chomitz and Kumari, 1998). This is partly a reflection of the diversity of conditions encountered: hydrological services, for example, depend on the rainfall regime, on the type of soil, and on topography. Deforestation can have multiple, often contradictory impacts, making the net impact on water services hard to determine. It can reduce infiltration, for example, but also reduce water use through evapotranspiration. The net impact of these changes (both in total and within a year) depends on the balance between these effects.

A related issue involves estimating how fast benefits would increase as a result of the proposed action (or decrease, if the action is harmful). Benefits obtained further in the future are less valuable. Another common mistake is to assume that the gain (or loss) of benefits would be more or less immediate. This is another area in which there is a need for reality checks. It is not uncommon to find older reports confidently predicting that under current deforestation trends, for example, a given area would have no trees left by a certain date—yet the date is not long past, and trees have not, in fact, vanished completely.

In contrast, the price side of the analysis tends to be much easier. Unless the change in the quantity of the service is very large (see Box 4.2 above), the unit value of the service is often unaffected. Thus if water flow to an irrigation system is reduced, and each additional liter of water available allows an increase in the value of production of US\$10, the main challenge is to estimate how much more (or less) water there will be. The value of production is determined by the interaction of supply and demand in a much wider market, and so will likely remain unchanged even if production in a given irrigated area falls.

There are some cases, however, in which the unit value of the service may change. If degradation reduces the quality of recreational opportunities, for example, it may not just lead to less visits (quantity change) but also to a lower willingness to pay per visit (price change).

As in the analysis of total flows, it is important to examine changes in net benefits. When considering the impact of changes, however, there are instances in which it can plausibly be supposed that the costs of service use do not change as a result of ecosystem degradation. In this case, the change in gross revenues alone would be sufficient to assess the change in benefits.

How do we use the results?

The main purpose of this type of analysis is to help guide decisionmaking by showing whether interventions are justified in economic terms: that is, whether the benefits resulting from the intervention exceed its costs.

If the sum of all the benefits from the proposed conservation intervention exceeds the sum of all its costs (including any opportunity costs of foregone benefits), the intervention would be beneficial from the point of view of society. That is, society as a whole would be better off, in economic terms, if the action were undertaken than if it were not. If costs exceed the benefits, on the other hand, the conclusion is that this conservation measure is not worth implementing, from an economic perspective.

Note that whether the result of the cost-benefit analysis is positive or negative, it only applies to

the specific conservation measure being considered in the analysis. Each possible conservation measure will have its own pattern of costs and benefits. Finding that one conservation measure is worth undertaking, in a specific case, does not mean that all conservation measures are worth undertaking. Nor does it mean that this same conservation measure is necessarily worth undertaking in a different situation. This is illustrated well in Case Study 4, which found that the same intervention (reforestation of burned areas) was very profitable in some areas and quite unprofitable in others.

Given that many benefits cannot be measured, estimates of benefits are often under-estimates. This is not a problem if the estimated benefits exceed the costs; measuring the other benefits would simply reinforce the conclusion that undertaking the proposed conservation intervention is desirable. This is illustrated in Case Study 5. For lack of data, many of the benefits of protecting Haiti's protected areas could not be estimated. Those benefits that could be estimated were already sufficient to justify the conservation measures, however.

When the estimated benefits of conservation are less than the estimated costs, on the other hand, a mechanical application of cost-benefit analysis would suggest rejecting the conservation option. It is possible, however, that conservation would have been accepted had all the benefits been measured. Some degree of judgment must enter at this point. By measuring at least some benefits, valuation can narrow the uncertainty over the net effect of the proposed intervention. This is also illustrated in Case Study 5: the minimum estimate,

of benefits, in the more pessimistic scenario of the conservation project's effectiveness, is less than the costs. The difference is very small, however (about US\$1 million, over a 50-year period), and it is quite likely that unquantified benefits would be sufficient to fill this gap, if it were possible to estimate them. Were the gap to be large, on the other hand, it would probably be wise to study the problem more carefully.

In some cases, a traditional benefit-cost analysis may not be feasible or desirable. For example, some ecosystems may be so unique that it might be felt they should be conserved at all costs. Valuation would still be useful by helping find the cheapest and most effective way of achieving the conservation objective.

Notes

- ¹ Barbier (1993, 1994) and others following him distinguish between 'impact analysis' which seeks to assess the damages inflicted on an ecosystem by a specific impact and 'partial valuation' which assesses project options. Here, we combine these two approaches into a single one that looks at how a change in management, whether deliberate or accidental, affects the flow of benefits provided by an ecosystem.
- ² The World Bank is conducting a series of studies of the cost of degradation in the Middle East and North Africa (Sarraf and others, 2004). However, these studies have so far focused on the impacts of pollution rather than loss of ecosystem services. Future studies will attempt to include loss of ecosystem services, to the extent that data allow.

6 Identifying winners and losers

The discussion thus far has focused on aggregate benefits and costs. In many cases, however, we are concerned not only about the magnitude of benefits, but also about who receives the benefits and who bears the costs. The third approach to valuation addresses this issue by attempting to identify who benefits from ecosystems, in what way, and how much.

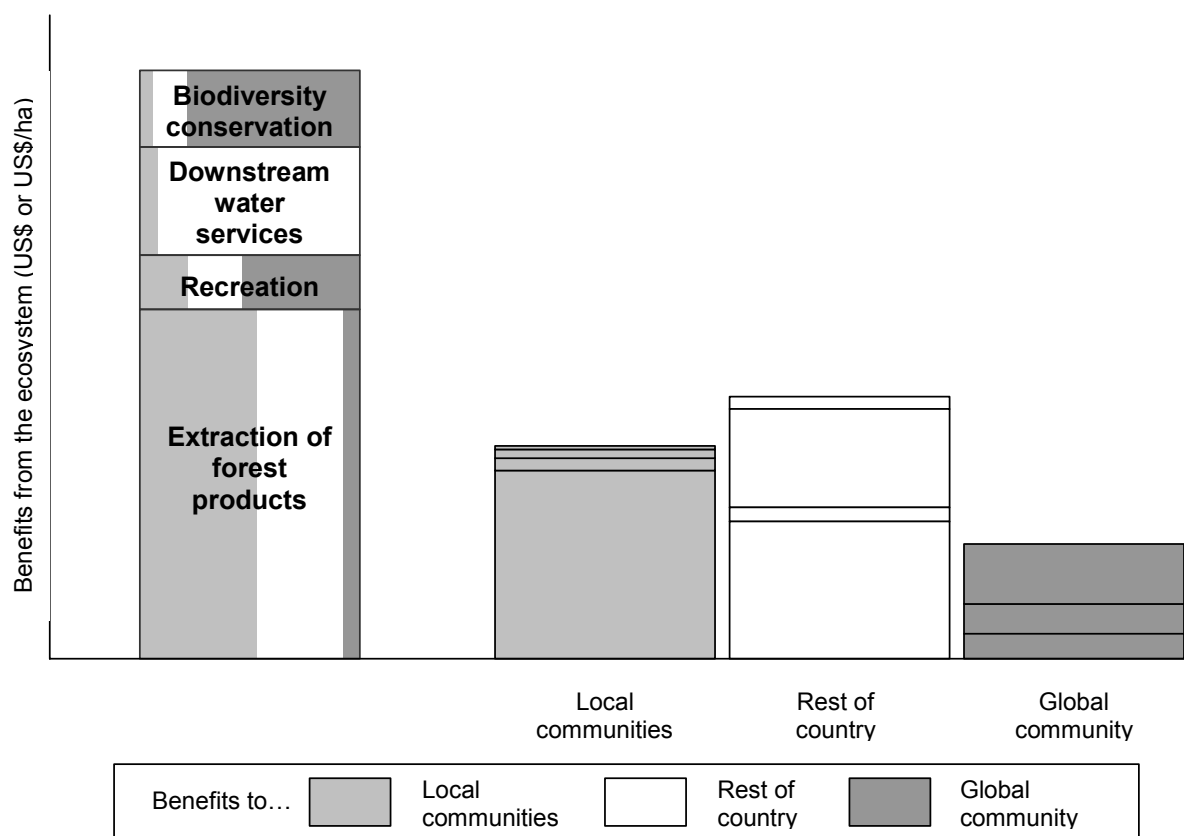
Why are we doing this?

If the increase in aggregate benefits exceeds the increase in aggregate costs, then conservation would be interpreted as being worthwhile from society's perspective. This is known as a 'Pareto improvement'—the benefits are sufficiently large that, in principle, everybody can be made better off (or, alternatively, that some can be made better off with no-one being worse off). But there is a difference between everyone being potentially better off, and everyone actually being better off. Consideration of aggregate benefits and costs masks the fact that those benefits and costs can be distributed very un-evenly across groups.

The un-even distribution of costs and benefits has both practical and ethical consequences. In practical terms, it is important to understand the costs and benefits received by local users, as they often have a very strong influence on how the ecosystem is managed. If local users stand to gain more from a particular land use, they may well convert the ecosystem to that land use no matter how large the benefits of conservation are to others. Likewise, if local users stand to benefit more from current conditions than from a proposed intervention, they are likely to oppose that intervention. Understanding who gains and—in particular—who loses from ecosystem conserva-

tion thus provides important insights into the incentives that different groups have to manage an ecosystem in a particular way. By comparing the net benefits that groups receive from an ecosystem managed in one way (without conservation, say) to the net benefits they would receive if it were managed in another way (with conservation), this approach can also help predict which groups are likely to support a change in management, and which groups are likely to oppose it. This approach can thus provide useful information in the design of appropriate responses.

More fundamentally, analysis of the distribution of costs and benefits is important to ensure that conservation interventions do not harm vulnerable people, and to design interventions that help reduce poverty and social exclusion. Tracking the flow of costs and benefits to different stakeholder groups allows us to understand how conservation actions affect the poor and other groups of interest, such as indigenous peoples. In the past, conservation efforts such as the creation of protected areas have often had a negative impact on many local communities, for example by reducing their access to resources upon which they depend for their livelihoods (see Case Study 9, for example). Such impacts are of greatest concern where the affected population is most deprived: even if the economic cost is small compared to the overall benefits, it could be very significant for poor households. Recent studies show that the poor are often very dependent on natural resources for their livelihoods (Cavendish, 2000; World Bank, 2003; Vedeld and others, 2004). They may well benefit, therefore, from healthier, more productive ecosystems. On the other hand, they may be harmed if access or use is restricted. Identifying and estimating the value of such

Figure 6.1: Distribution of ecosystem benefits

impacts can allow for conservation strategies to be modified to avoid them, for appropriate compensation mechanisms to be designed, or for financing schemes to be developed (as discussed in the next chapter).

How do we do it?

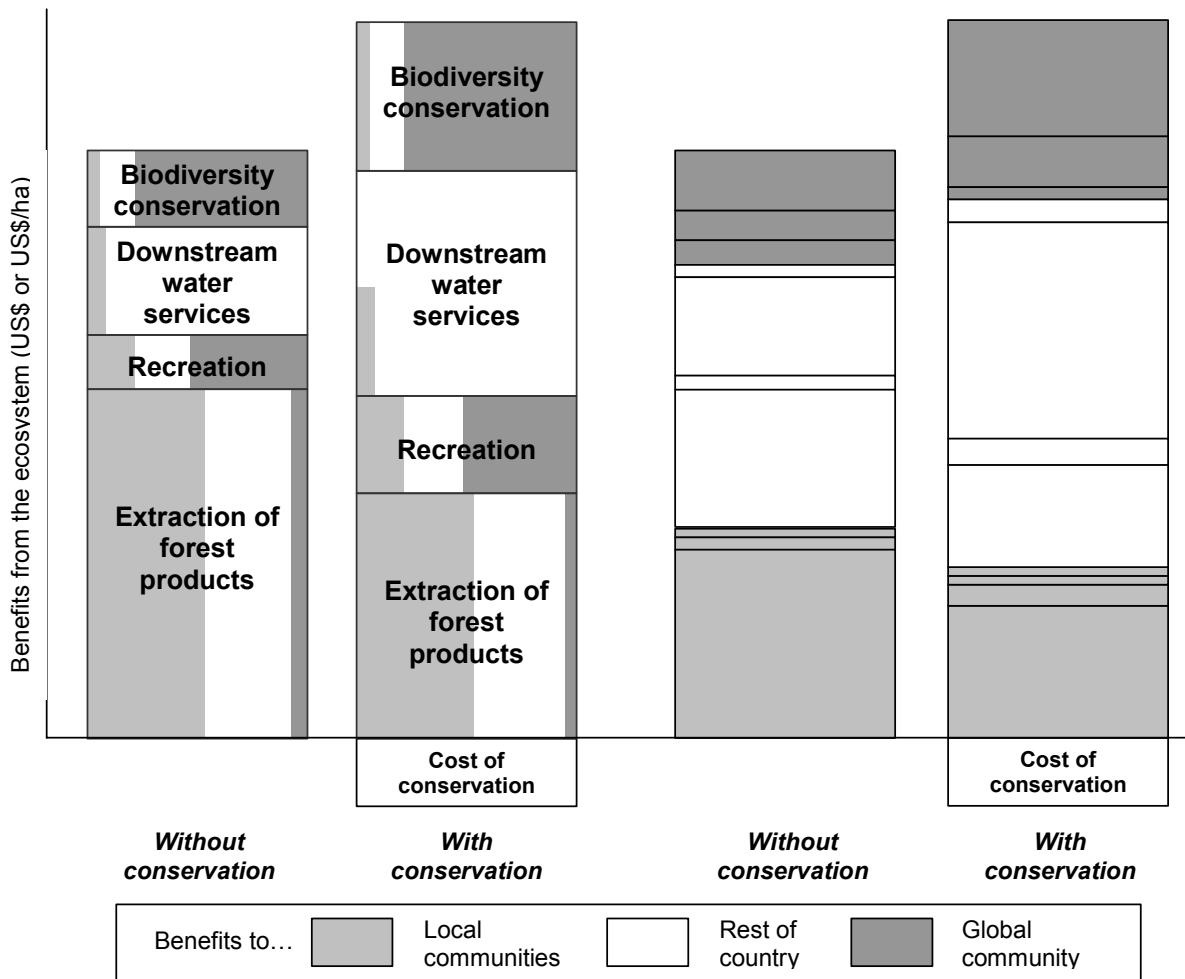
Valuing the benefits and costs obtained by individual groups obviously requires identifying the groups of interest and the specific services they use. An initial breakdown that is useful in many cases is between local communities (who often receive the bulk of direct use values), the rest of the nation (who receive some direct use values, such as recreation, but typically receive the bulk of indirect use values), and the rest of humanity (who should be distinguished from the rest of the nation if the analysis is undertaken from a national perspective). In many cases, of course, it is necessary to subdivide groups much more finely: rubber tappers and loggers may both be local, for example, but they derive very different benefits from a forest and thus have different

interests. Likewise, people who live downstream of an ecosystem, in the same watershed, stand to benefit from water regulation services; people who do not live there may receive other benefits, but they do not receive water regulation services (Pagiola and Platais, forthcoming). The simplest sub-division, of course, is into service users: water users, recreational users, and so on. Though crude, this can sometimes be useful. Most often, however, multiple groups share the same benefit, and most groups get more than one benefit.

This approach is illustrated in Figure 6.1, in which the total flow of benefits from an ecosystem is broken down into three groups: those received by local users, those received by the rest of the country, and those received by the global community. A similar analysis could be conducted showing how benefits would change as a result of a conservation intervention; this is illustrated in Figure 6.2.

Once individual groups have been identified, the analysis proceeds as described in Chapters 4 and

Figure 6.2: Effect of a conservation intervention on the distribution of ecosystem benefits



5. Indeed, the analysis described in Chapters 4 and 5 can be undertaken by estimating the benefits and costs received by different groups, and then aggregating up. As noted previously, it is important to consider not just direct costs and benefits, but also the opportunity costs that groups may face if they are prevented from undertaking certain uses of ecosystems.

As already noted in Chapter 4, when the analysis is undertaken from the perspective of individual groups, then it should include any taxes that the group pays, or subsidies that it receives. If the analysis is being undertaken in order to understand a group's incentive to conserve, for example, retaining distortions is important because they affect how the group perceives net benefits and costs. For example, if fuels are taxed, then extraction of fuelwood from natural eco-

systems will be perceived as relatively valuable, and groups that use fuelwood will have a greater interest in conserving ecosystems that provide it. Conversely, if fuels are subsidized, then this direct use service will not be perceived as being very valuable, and there will be little interest in protecting the ecosystem that provides it. Likewise, if the analysis is being undertaken to understand how a group's livelihood depends on ecosystem services, retaining the distortions is appropriate because they affect how valuable those services are to that group. When the analysis is undertaken from the perspective of society as a whole, on the other hand, it is appropriate to correct for the effect of distortions (see Box 6.1).

It is often important to understand not only the absolute amount of benefit a given group may receive, but also what role that benefit plays in the

group's livelihood strategy. A benefit may appear small in absolute terms, for example, but play an important role because it acts as a safety net when other income sources are unavailable.

Box 6.1: Financial vs economic analysis

Valuation can be carried out either from the perspective of society as a whole ('social' or 'economic' analysis) or from that of individual groups within society ('private' or 'financial' analysis) (Gittinger, 1982; Monke and Pearson, 1989). Focusing on a particular group usually requires focusing on a subset of the benefits provided by an ecosystem, as that group may receive some benefits but not others. It will often also require using estimates of value specific to that group; the value of additional water, for example, will be different depending on whether it is used for human consumption or for irrigation. It should also include any taxes that the group pays, or subsidies that it receives (directly or indirectly). When the analysis is undertaken from the perspective of society, however, the social opportunity costs should be used to value resources. In particular, these values should *not* include any taxes or subsidies, as these are simply transfers between different groups.

How do we use the results?

Case Study 8 illustrates the results of one such analysis of the costs and benefits of Madagascar's

protected area system. Overall, this system provides net benefits to the country, thanks to the valuable watershed protection services these areas provide, their tourism benefits, and payments received from the global community for protection of the country's unique biodiversity. But these benefits are very unevenly distributed. Local communities bear the brunt of the costs, as they are barred from using protected areas either for agriculture or for the collection of fuelwood and other non-timber forest products (NTFPs). Downstream water users such as irrigated farmers benefit substantially, as do tourism operators. The protected area management agency, ANGAP, bears the management costs but receives external support (and a part of the tourism benefits). These results indicated the need for support to protected areas to include appropriate compensation mechanisms for local communities, and such mechanisms were included in a project to support the country's protected area system.

The other major purpose of this type of analysis, as discussed further in the next chapter, is to identify those who benefit from ecosystem conservation—either in the country itself or outside it. This can help to identify potential financing sources for conservation.

7 Identifying potential financing sources

Conservation can bring high benefits, but only if it takes place. Often, it does not, for lack of resources. Cash-strapped governments are often very reluctant to spend on conservation. The fourth approach is aimed at assessing the potential financing sources to pay for conservation.

Why are we doing this?

Effective conservation usually requires a long-term commitment of resources.

Financing conservation has two dimensions. One is to secure sufficient resources, at any given time, to cover the costs of conservation. Almost invariably, the resources available for conservation are grossly inadequate to the task. Thus, even if conservation could, in principle, generate large economic benefits, it often does not happen. Or, more commonly, it happens for some time, thanks to funding from a donor, and then collapses once the project and its funding come to an end.

The second dimension is to make conservation as financially self-sustaining as possible. Budget constrained governments often balk at devoting significant resource to conservation even if the benefits of doing so are clear. Moreover, budget shortfalls and other problems may well curtail future funding even if the benefits of conservation are well understood. Accordingly, there have been increasing efforts to develop mechanisms to ensure that conservation is, as much as possible, self-financing, so that it is not held hostage by the annual vagaries of government budgeting decisions and donor aid allocations. These efforts range from traditional approaches such as charging entry fees to visitors to protected areas to

innovative approaches such as payments for environmental services (PES).

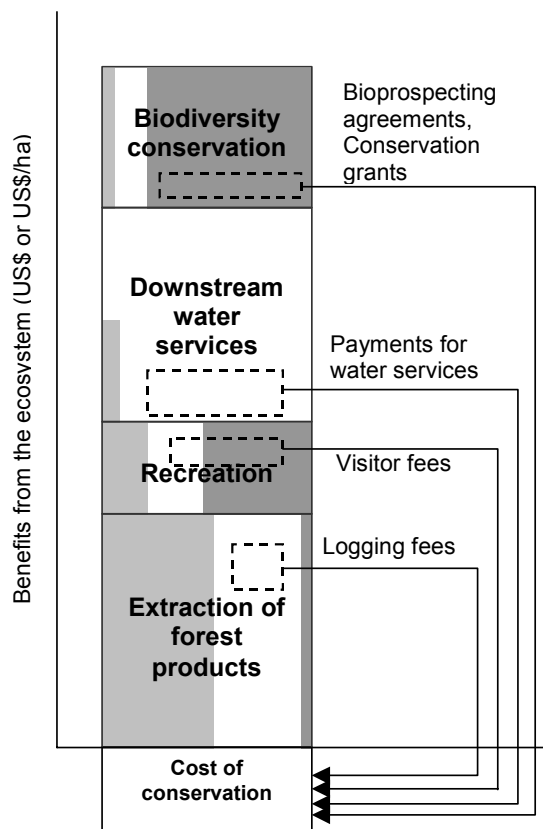
Economic valuation can help make conservation financially sustainable in two ways. First, by demonstrating the benefits that ecosystems generate, and the increased benefits (or avoided losses) that conserving these ecosystems can bring to stakeholders, valuation can help convince decisionmakers to allocate more resources to conservation. The analysis of the benefits of Madagascar's protected areas, described in Case Study 8, was instrumental in building support for a project to strengthen the country's protected areas system. But it would be overly optimistic to expect all problems to be resolved this way.

Second, valuation can provide invaluable support to these efforts by identifying and quantifying the major benefits provided by a given ecosystem, and by identifying the beneficiaries. Based on this information, a variety of approaches might be used to secure additional funding for ecosystem conservation. Several of these approaches could potentially generate funds that would go straight to conservation, thus helping make it self-sustaining.

How do we do it?

The first step involves identifying the financing needs. As discussed earlier, these involve two sets of costs: the actual out-of-pocket cost of conservation (for example, paying park rangers) and the foregone benefits resulting from restrictions on certain kinds of uses of the ecosystems being protected. This second category of costs is not a financial cost to the conservation agency. It can become a financial cost, however, if the affected

Figure 7.1: Identifying potential financing sources for conservation



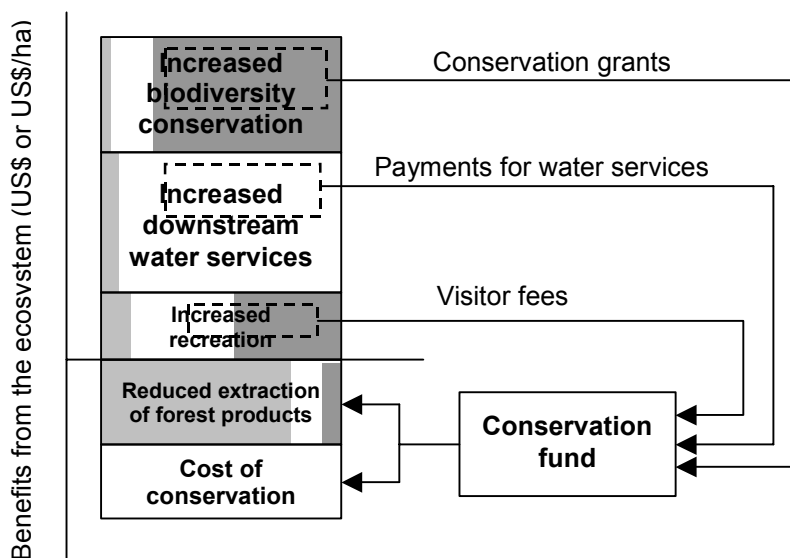
serve or for equity reasons. Many countries, including Bolivia, Madagascar, and Costa Rica, have adopted policies of compensating affected stakeholders.

The next step, as in Chapter 6, is to identify the beneficiaries of each service an ecosystem is providing. As these groups are benefiting from the ecosystem, it is in their interest to contribute to conserving it. Different mechanisms might be used to capture some of the benefits these groups are receiving, so as to make them available for conservation. This approach is illustrated in Figure 7.1.

For some types of services, it is often politically much easier to charge service users when a change is involved. This is particularly true of indirect use values. Service users often balk at paying for services they are already receiving for free, even when they benefit handsomely from them. It is often easier to convince them to pay when changes in benefits are involved: an increase in benefits, or an avoided loss of a benefit. Likewise, some donors will only finance activities that bring incremental gains. The analysis would be similar, but be based on examining the breakdown of benefit changes from a given conservation intervention, as in Figure 6.2. This approach is illustrated in Figure 7.2.

stakeholders need to be compensated for their losses, either to change their incentives to con-

Figure 7.2: Identifying potential financing sources for a conservation intervention



How do we use the results?

When recreational use is important, there is often a potential to use entrance fees. It is important to know to what degree recreational use is undertaken by foreign visitors as opposed to national visitors, as this may affect viable fee levels. Such mechanisms are already widely used, although fee levels are typically set far below their potential. A review of the economics valuation literature finds that foreign visitors are willing to pay considerably higher amounts than the fees currently charged for visits at developing country natural areas (Lindberg and Aylward, 1999). Some protected areas systems already generate substantial amounts of resources in this way. The South African National Park System, for example, recovers 80 percent of its budget costs from fees and tourism business it operates in parks (Eagles, 2001). This approach is only likely to work when access to conserved areas can be controlled.

Some extractive uses are also susceptible to being tapped for increased funding. It may be difficult to charge local users for collecting fuelwood and NTFPs, but relatively easy to increase royalties paid by loggers.

When indirect uses such as watershed protection provide important benefits, then payments for environmental services provide a promising approach. In a PES program, downstream water users pay fees which are used to finance payments to land users in upper watersheds who undertake appropriate land uses. Several cities and towns have implemented such programs (Box 7.1). Other PES programs focus on carbon sequestration services. Biodiversity benefits are often the hardest to capture, but even here there has been considerable experimentation with a variety of

approaches. Note that it is often unrealistic to expect to capture the entire benefit from various user groups. As illustrated in Figure 7.1, typically only a portion of their benefit can be captured. (see Box 7.1 and Case Study 10).

Box 7.1: Paying for watershed protection

Recent years have seen an increasing use of mechanisms based on the principles of payments for environmental services (PES), particularly in Latin America. Costa Rica and Mexico have created nationwide PES programs. The vast majority of PES initiatives, however, have been for smaller-scale initiatives at the scale of individual watersheds. Irrigation water user groups, municipal water supply systems, and hydroelectric power producers in several countries participate in such programs. The cities and towns that use PES to protect their water supplies cover a wide spectrum, from Quito, Ecuador, with 1.2 million people, to Yamabal, El Salvador, with only 3,800 people.

Valuation is a critical step in the development of PES programs. The payments must obviously exceed the additional benefit to land users of the alternative land use (or they would not change their behavior) and less than the value of the benefit to downstream populations (or they would not be willing to pay for it). Without valuation, it may be difficult to set an appropriate payment level, or even to determine whether the program is worth implementing at all.

Source: Pagiola and Platais, forthcoming.

Useful to know elasticity of responses to price changes. Attempts to charge high prices may drive off visitors, for example, defeating the aim of generating increased revenue. On the other hand, reducing visitor numbers may be desirable, to reduce pressure. Knowing the elasticity helps predict how visitor numbers and revenue would change as the price is changed.

8 Conclusions

Only a few decades ago, many standard economics textbooks considered environmental costs and benefits to be ‘unquantifiable’ and advised against trying to estimate them. Since then, tremendous progress has been made in developing a range of techniques for valuing environmental costs and benefits. Today our toolkit is well-stocked with increasingly sensitive and generally reliable valuation methods. There is a growing body of literature that applies these techniques to a wide range of environmental issues. There is no longer any excuse for considering environmental costs and benefits as unquantifiable.

The potential of economic valuation

Economic valuation can provide useful information—when it is done correctly. Valuation of ecosystem goods and services can seem deceptively simple—a matter of multiplying a price by a

quantity. In practice, however, valuation is often complex and normally requires specialist training and experience to ensure credible results. But even expert economists will produce information that is of little use if the questions they are asked are badly framed. As this paper has sought to clarify, different policy contexts require different approaches. In particular, estimates of the total annual flow of benefits from an ecosystem, while often very impressive, are a poor guide to policy and investment decisions. Much more useful in most cases are estimates of the changes in benefit flows that will result from changes in ecosystem management. Table 8.1 summarizes the main characteristics of each approach.

The four approaches described here are closely linked and build on each other. They represent four different ways to look at similar data regarding the value of an ecosystem: its total

Table 8.1: Approaches to valuation

| <i>Approach</i> | <i>Why do we do it?</i> | <i>How do we do it?</i> |
|---|--|--|
| Determining the total value of the current flow of benefits from an ecosystem | To understand the contribution that ecosystems make to society | Identify all mutually-compatible services provided; measure the quantity of each service provided; multiply by the value of each service |
| Determining the net benefits of an intervention that alters ecosystem conditions | To assess whether the intervention is economically worthwhile | Measure how the quantity of each service would <i>change</i> as a result of the intervention, as compared to their quantity without the intervention; multiply by the marginal value of each service |
| Examining how the costs and benefits of an ecosystem (or an intervention) are distributed | To identify winners and losers, for equity and practical reasons | Identify relevant stakeholder groups; determine which specific services they use and the value of those services to that group (or changes in values resulting from an intervention) |
| Identifying potential financing sources for conservation | To help make conservation financially sustainable | Identify groups that receive large benefit flows, from which funds could be extracted using various mechanisms |

value or contribution to society, the change in this value if a conservation action is undertaken, how this change affects different stakeholders—that is, who are the beneficiaries and who are the losers—and how beneficiaries could be made to pay for the services they receive to ensure that the ecosystem is conserved and its services are sustained. Each of these approaches to valuation uses similar data. They use that data in very different ways, however, sometimes looking at all of it, sometimes at a subset, sometimes looking at a snapshot, and sometimes looking at changes over time. Each approach has its uses and its limitations. Understanding under what conditions one should be used rather than another is critical: the answer obtained under one approach, no matter how well conducted, is generally meaningless when applied to problems that are better treated using another approach. In particular, using estimates of total flows to justify specific conservation decisions—although commonly done—is almost always wrong. Properly used, however, valuation can provide invaluable insights into conservation issues.

Economic analysis is not and should not be the only input into conservation decisions. People can and do decide to conserve things based on a range of other criteria, such as for ethical, cultural, and historical reasons. Even then, valuation can provide relevant information—for example, by highlighting the economic consequences of alternative courses of action. Thus economic valuation, used correctly, will lead to more informed choices even when economic considerations are not the primary criterion for decisionmaking.

It is rarely feasible or desirable to estimate every environmental benefit or cost. Even where valuation provides only partial results, however, it can help to structure how we think about conservation, identify critical information gaps, and clarify the relation between ecosystem processes and human welfare. Indeed, an important benefit of attempting to undertake economic analysis is that it forces us to grapple with our limited understanding of ecosystem processes and the way they affect human welfare. All too often, public debate and policy on conservation is based on vague statements about ecosystems benefits, which implicitly assign a value of either zero or infinity to natural

ecosystems. Zero is clearly wrong, but infinity is equally unhelpful as it prevents us from setting priorities. The types of analyses discussed in this paper force us to be explicit about our assumptions: what specific services does an ecosystem provide? Who receives those services? How important are they? How would each of these services change if the ecosystem were managed differently? How big would the change be? How rapid? How long-lasting? Would it be reversible? What substitutes exist, if any? Simply stating the questions involved in an economic valuation can help to identify what we know and what we don't know about the role that ecosystems play in our welfare.

Common pitfalls of valuation

With their hundreds of pages, tidy tables, colorful figures, and glossy covers, economic valuation reports often look most impressive. But are they any good? In this paper, we have emphasized the need to frame the question appropriately to ensure that the valuation provides answers that are useful and relevant for the policy issue at hand. There are, of course, also many other ways in which valuation can go wrong. Although it is not the objective of this paper to provide detailed instructions on how to undertake valuation, the discussion in the preceding chapters has brought out several pitfalls often encountered when doing so. Table 8.2 summarizes some of the key messages. This is by no means a comprehensive list of potential pitfalls, but it highlights some of the most important.

The limits of economic valuation

While valuation can shed useful light on many issues, there are several questions that economic valuation techniques handle poorly. Most of the direct and indirect use values of ecosystems can be measured quite accurately and reliably—the main constraint is often the availability of relevant physical data (that is, information on the quantity of service provided, or on the change in the quantity of service provided) rather than economic data (on the value of an extra unit of the service). Estimating option values and existence values involves greater uncertainties. Valuation of changes in human mortality is also problematic, as many people (including many economists!) find the

Table 8.2: Avoiding common pitfalls to valuation

| | |
|---|--|
| ▪ Use net benefits, not gross benefits | Failing to consider the costs involved in using resources (the cost of harvesting products, for example, or the cost of piping water from its source to the user) results in an over-estimate of the value of ecosystem services. |
| ▪ Include opportunity costs | The cost of an action are not limited to the out-of-pocket costs involved in implementing it. They also include the opportunity costs resulting from the foregone benefits of alternative actions (or inaction). Omitting opportunity costs makes actions seem much more attractive than they really are. |
| ▪ Don't use replacement costs | ... unless you can demonstrate (i) that the replacement service is equivalent in quality and magnitude to the ecosystem service being valued; (ii) that the replacement is the least cost way of replacing the service; and (iii) that people would actually be willing to pay the replacement cost to obtain the service. |
| ▪ Don't use benefits transfer | ... unless the context of the original valuation is extremely similar to the context you are interested in. Even then, proceed with caution. However, it is a good idea to compare your results to those obtained elsewhere. |
| ▪ Don't use value estimates based on small changes in service availability to assess the consequences of large changes in service availability | Economic value estimates are not independent of the scale of the analysis. Value estimates are almost always made for small ('marginal') changes in service availability, and should not be used when contemplating large changes. |
| ▪ Be careful about double-counting | Many valuation techniques measure the same thing in different ways. For example, the value of clean water might be measured by the avoided health care costs or by a survey of consumer WTP for clean water. But consumer WTP for clean water is due (at least in part) to their desire not to fall sick, so these two results should not be added together. If they are, the value of clean water will be over-estimated. |
| ▪ Don't include global benefits when the analysis is from a national perspective | More generally, only consider benefits (or costs) that affect the group from whose perspective the analysis is being undertaken. Including benefits which are primarily global in nature in an analysis undertaken from a national perspective is a particularly common form for this mistake, and results in an over-estimate of the benefits to the country. |
| ▪ Adjust for price distortions | ... when conducting the analysis from the perspective of society as a whole, but not when conducting the analysis from the perspective of an individual group. |
| ▪ Avoid spurious precision | Most estimates are by necessity approximate. Don't simply paste the result in the spreadsheet, with its three decimal points, into the report: round the result appropriately. When there is substantial uncertainty, report the results as ranges. |
| ▪ Submit results to sanity checks | Are the results consistent with other results? Are they reasonable in light of the context? Extraordinary results are not necessarily wrong, but must be checked carefully. Extraordinary results require extraordinary proof. |

notion of assigning a monetary value to human life unacceptable.

Economic valuation also tends to handle very large-scale and long-term problems rather poorly.

Existing economic valuation techniques can provide reliable answers to questions involving relatively small-scale changes in resource use or availability, but become less robust as the scale of the analysis and the magnitude of environmental change increases. Similarly, economic valuation tends to deal poorly with very long time horizons. Uncertainty about future benefit flows becomes more and more important, and the role of discounting increasingly determinant. Alternative approaches such as the Safe Minimum Standard (SMS) approach may be more suitable in such

cases, particularly when changes are thought to be irreversible (Bishop 1978; Crowards, 1998).

Economic valuation has both strengths and limitations as a tool for decisionmaking. It is clear, however, that decisions about environmental management are not getting easier, and that information about costs and benefits is increasingly essential to ensure efficient, equitable, and sustainable outcomes. Valuation can play an important role in providing such information, provided it is used correctly.

9. Case studies

This section provides detailed summaries of several valuation studies, illustrating the various approaches described in this paper. They are drawn from a range of situations in a variety of countries. Some reflect sophisticated analyses undertaken with abundant data, while others had

to make do with limited data of uncertain reliability.

A companion CD-ROM to this paper provides additional readings and case studies.

Table 9.1: Case studies of economic valuation of ecosystem conservation

| <i>Study</i> | <i>Type of study</i> | | | |
|---|--|--|---------------------------------|--------------------------|
| | <i>Value of total flow of benefits</i> | <i>Value of change in flow of benefits</i> | <i>Distribution of benefits</i> | <i>Financing options</i> |
| 1. The total economic value of Mediterranean forests | • | | | |
| 2. The value of natural capital in Sub-Saharan Africa | • | | | |
| 3. The impact of deforestation on Ghana's national savings rate | • | | | |
| 4. The benefits of reforestation in coastal Croatia | | • | | |
| 5. The benefits of protecting Haiti's forest remnants | | • | | |
| 6. The value of mangrove forests as fish nurseries in Thailand | | • | | |
| 7. Tourism vs logging in Palawan | | • | | |
| 8. The costs and benefits of Madagascar's protected areas systems | • | | • | |
| 9. The impact of conservation on local communities in Uganda | | | • | |
| 10. Paying for water services in New York State | | | | • |

Case study 1: The Total Economic Value of Mediterranean forests

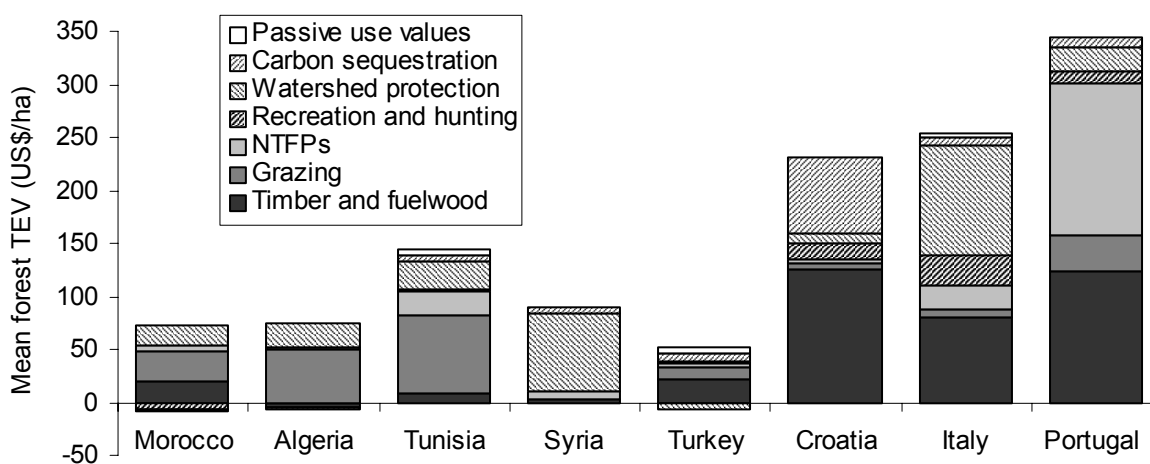
On-going deforestation in many parts of the world has stimulated interest in estimating the benefits of forests, partly to identify the values being lost but mainly to justify conservation efforts. The figure below summarizes the estimated benefits of forests in several Mediterranean countries. The average TEV of forests in the eighteen countries studied is about US\$150/ha a year. This is likely to be an underestimate, however, as many non-market benefits could not be estimated in many cases. The gap between the estimated TEV in European countries and that in North African and Middle Eastern countries is probably smaller than it appears here, as data constraints were particularly severe in the latter countries.

Direct use values contribute about 65 percent of the estimated TEV, although this share is likely over-estimated as it is easier to measure direct uses than other values. Timber and fuelwood generally account for less than a third of estimated TEV, on average. In North African countries, the importance of timber and fuelwood is dwarfed by the value of grazing. Cork drives up the contribution of NTFPs in Portugal. Recreation and hunting benefits were imperfectly measured, but in Euro-

pean countries these benefits rival and sometimes exceed timber values. Watershed protection is an important benefit in Italy, Syria, and the three *Maghreb* countries, and would likely have played an important role in several other countries as well, had it been possible to better estimate its value. Carbon sequestration provides relatively low benefits, and is negative in countries like Morocco where on-going deforestation means that forests are net sources of emissions. (Note that carbon is valued at its estimated benefit to global society, using Fankhauser’s (1995) estimate of US\$20/ton carbon, not at its benefit to the individual countries). The estimates for passive use values are scarce and partial.

On a per capita basis, forests in Mediterranean countries provide at least US\$50 annually—about US\$70 per capita in European countries, but less than US\$10 in North African and Middle Eastern countries, with their smaller forest areas. As these are estimates of annual benefit flows obtained from forests, it is legitimate to compare them to other flows, such as GDP. On average, they amount to about 1 percent of GDP

Annual flow of benefits from forests in Mediterranean countries



Source: from data in Croitoru and Merlo (forthcoming).

Case study 2: The value of natural capital in Sub-Saharan Africa

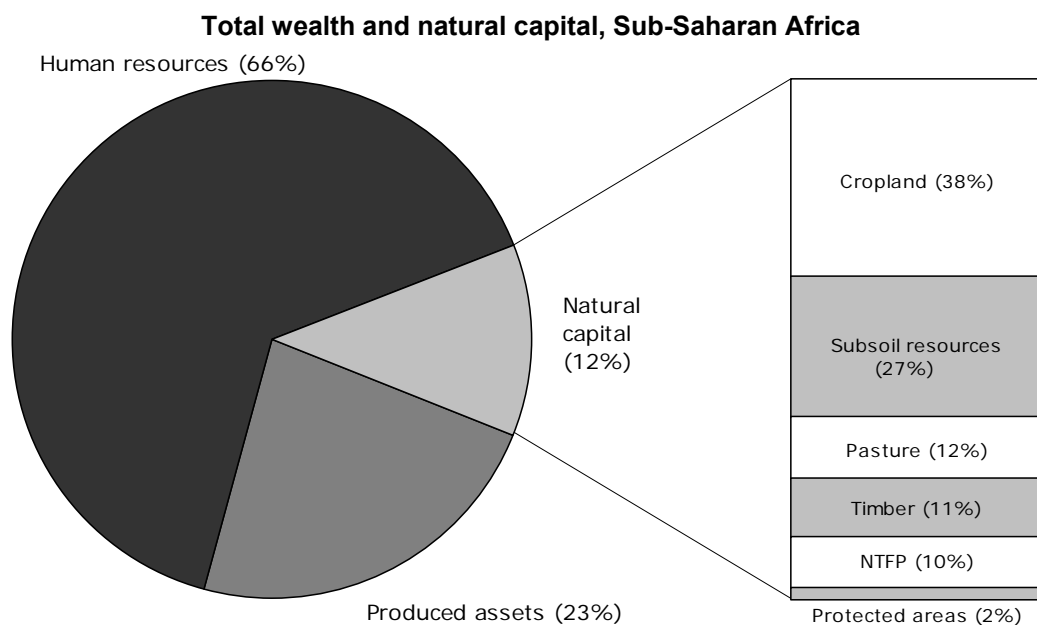
Ecosystems are often described as forming part of 'natural capital'. This analogy suggests several questions: how does one measure its value? And how important is natural capital as a source of wealth, compared to other forms of capital?

The economic value of an asset is usually taken to be the discounted sum of all current and future benefits it will generate. This approach can be applied to natural capital: by estimating the flow of current and future benefits that an ecosystem will generate, it is possible to assess its capital value.

The figure below shows the estimated value of natural capital in sub-Saharan Africa. These estimates are based on the current and future benefits generated from sub-soil reserves; pasture land; crop land; and forests. In particular, the capital value of Africa's forest stock is calculated from the estimated sustainable flow of timber production, NTFPs, and the benefits from protected forests. The annual flow of round wood production is multiplied by the average timber rent (that is, the difference between the market price and average unit cost of production). The value of NTFPs is based on estimates obtained for both

developed and developing countries (Lampietti and Dixon, 1995). The value of protected areas is based the opportunity cost of foregone benefits from converting them to pasture or agricultural land. This may be considered a lower bound estimate.

The results suggest that sub-Saharan Africa, like other regions, relies above all on its people as the basis of its future welfare. Human resources (human capital plus raw labor) alone account for almost two-thirds of the region's total wealth. Produced assets such as roads, bridges, and buildings are a much smaller share. Natural wealth constitutes 12 percent of the total. Roughly half of the latter figure reflects the value of cropland and pasture lands. The implication for policy is the importance of judicious land management. Forests are also very important, accounting for nearly one quarter of total natural wealth. These data tell us that Africa's future welfare heavily depends on the ability of its people to manage, among other things, the stock of natural soil fertility and natural forests. (Sub-soil resources are important in the aggregate, but are very unevenly distributed among countries.)



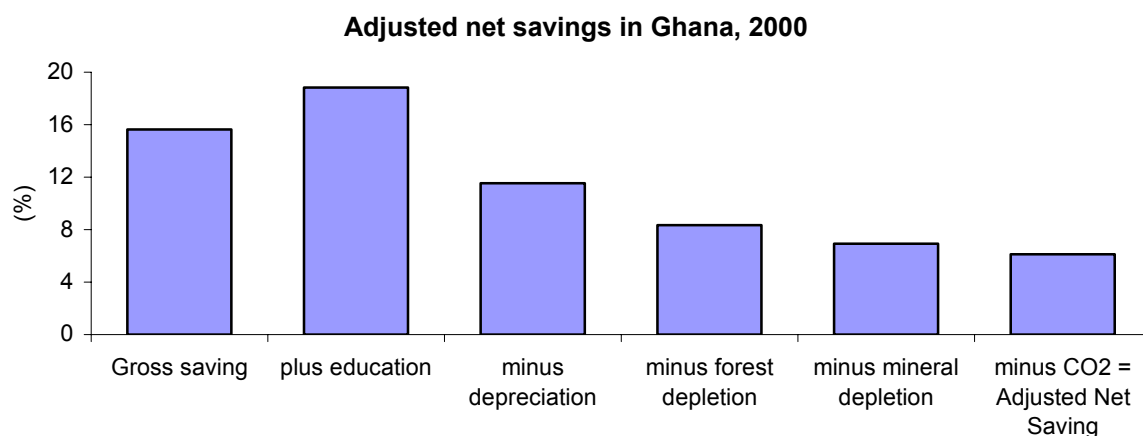
Source: Kunte and others (1998) and World Bank staff estimates.

Case study 3: The impact of deforestation on Ghana’s national savings rate

Some economists define sustainable development as a process characterized by a non-declining per capita welfare (or utility) over time. Maintaining positive changes in real wealth (‘genuine’ or ‘adjusted net’ saving) is a necessary condition to achieve sustainable development.

The World Bank’s estimates of Adjusted Net Savings (ANS) measure the change in total wealth in a given period. ANS is calculated by adjusting the traditional measure of net national savings to account for activities which enhance wealth, such as education expenditure (an investment in human capital), as well as activities that reduce wealth, such as depletion of mineral and energy reserves, forest depletion, damages from carbon dioxide and health damages from particulates. The World Bank is working on extending ANS to include changes in other types of natural capital, but data availability is an important constraint.

The figure below displays these adjustments to saving in the case of Ghana in 2000. The first column is the traditional national accounts measure of gross saving, Gross National Income (GNI) minus consumption. Successive columns then add or subtract values in order to arrive at the ‘bottom line’ measure. The adjustments are considerable—whereas the Ghanaian Minister of Finance presumably thinks that the saving rate is over 15 percent of GNI, in fact the true rate of saving is only about 6 percent. Forest depletion accounts for about 3 percentage points of this adjustment. Note that, for lack of data, the adjustment here is only for depletion of forest (that is, the amount by which forest harvest exceeds natural regeneration). Had it been possible to estimate the degradation of other goods and services provided by natural ecosystems, the estimated ANS would likely have been even lower.



Sources: World Bank (2004); Hamilton and Clemens (1999).

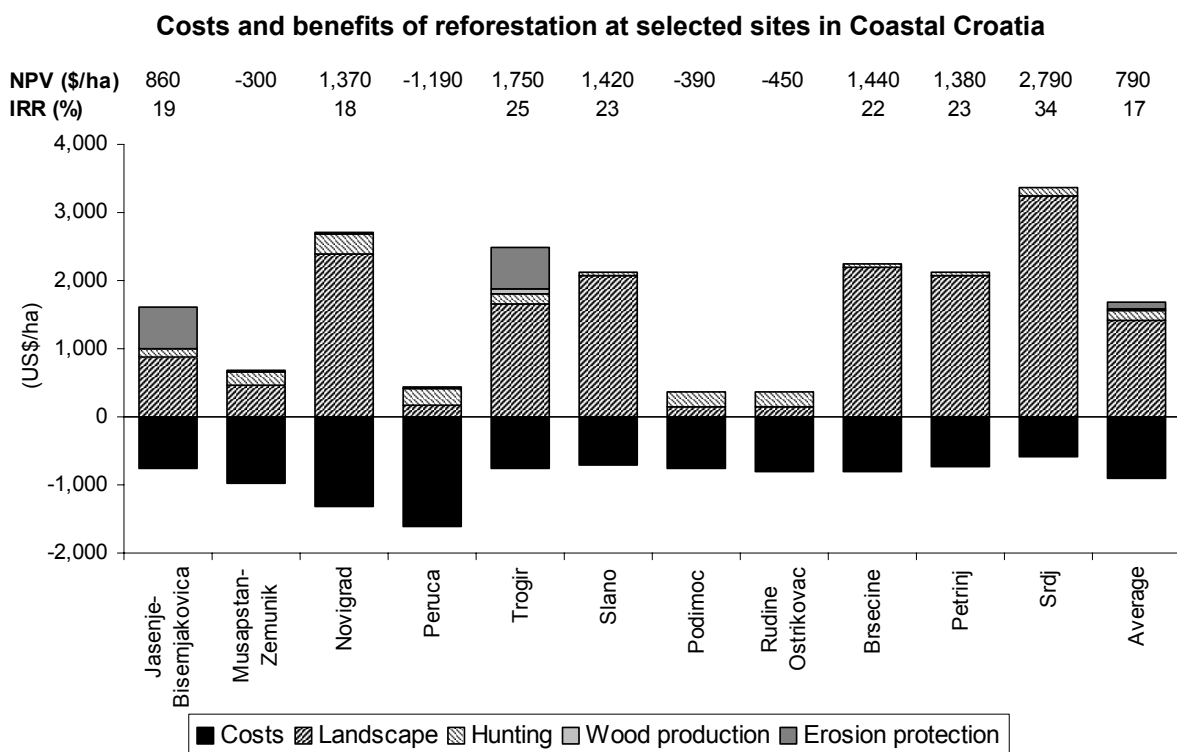
Case study 4: The benefits of reforestation in coastal Croatia

Croatia's coastal forests play an important role in the country's tourist industry, as they are a key element in the landscape. The *Croatia Coastal Forest Reconstruction and Protection Project*, which was financed by the World Bank, included reforestation of several forest areas which had been damaged by fire.

The appropriate question in this case was whether the additional benefits obtained by reforestation, compared to allowing natural regeneration to occur, justified the additional costs. In each case, the analysis centered on (i) the degree to which specific benefits would recover with reforestation; and (ii) the increased rate at which benefits would recover.

The figure below shows the results for each proposed reforestation site: in each case the expected benefits are compared to the expected costs. Both

vary by site, depending on local characteristics. Sites on steeper slopes, for example, are costlier to reforest. The overall analysis, shown in the last column, indicates that this component, as designed, is beneficial, with a NPV of US\$790/ha (discounted at 10 percent), and an internal rate of return (IRR) of 17 percent. However, this overall result masks the fact that the net benefits of reforestation vary substantially from site to site—even within the same county, benefits can vary by several orders of magnitude. If only the sites with positive net benefits are included, the average benefits almost double to US\$1,570/ha, and the overall IRR rises to 24 percent. As a result of this analysis, the component was restructured to drop all the proposed sites that were found to have negative net benefits, and guidelines were developed, based on the characteristics of the high-benefit sites, to select additional sites to meet the original reforestation target.



Source: World Bank, 1996a.

Case study 5: The benefits of protecting Haiti’s forest remnants

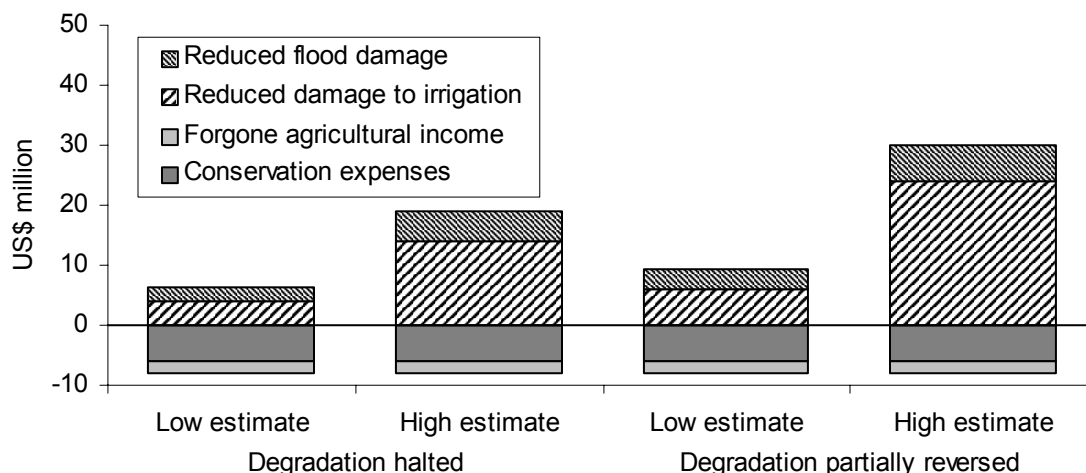
With a per capita annual income US\$440, Haiti is the poorest country in the western hemisphere. It also scores extremely low on a wide range of indicators such as literacy, child mortality, and life expectancy. Does it make sense for such a country to devote resources to protected areas? An analysis prepared for the Haiti Forest and Parks Protection Technical Assistance Project, which was financed by the World Bank, indicates that it does.

In the presence of substantial data constraints, this analysis used order-of-magnitude estimates to predict the impact of continued degradation of the remaining forest areas. These forests are upstream of important irrigated areas, which have been substantially affected by siltation caused by prior deforestation. The impact of additional deforestation was predicted by extrapolating the impact of previous deforestation. The resulting reduction in irrigated agricultural production was then estimated (an application of the *production function* approach). These estimates were made under two scenarios: (i) that the project would halt degradation at current levels, and (ii) that it would partially reverse past degradation. Similar estimates were made for damage to infrastructure such as roads. Other impacts were only assessed quali-

tatively, for lack of data. These expected benefits were then compared against the expected costs of protecting the targeted protected areas, including actual conservation costs under the project and beyond, and the opportunity cost of not logging the area and converting it to agricultural production.

The results are presented below. Given the weakness of the data, results are shown as broad ranges rather than single figures, with high and low estimates for each scenario. These results suggest that the benefits of conservation will exceed the costs: the avoided losses to downstream irrigated agriculture, for example, far exceed the foregone gains from converting upstream areas to agriculture, even if the project succeeds only in halting degradation at current levels. Gains to downstream producers would be even greater should the project succeed in partially reversing degradation. In short, it seems likely that maintaining these protected areas is in the country’s own interest. These results are strengthened when it is noted that many potential benefits could not be estimated and were omitted from the numerical results. Global benefits, while not quantified, further reinforce this conclusion.

Estimated costs and benefits of natural reserve management in Haiti



Notes: Only quantified costs and benefits shown; numerous other benefits remain unquantified. Quantified costs and benefits shown in present value terms, discounted at 10 percent over an infinite time horizon

Source: World Bank, 1996b.

Case study 6: The value of mangrove forests as fish nurseries in Thailand

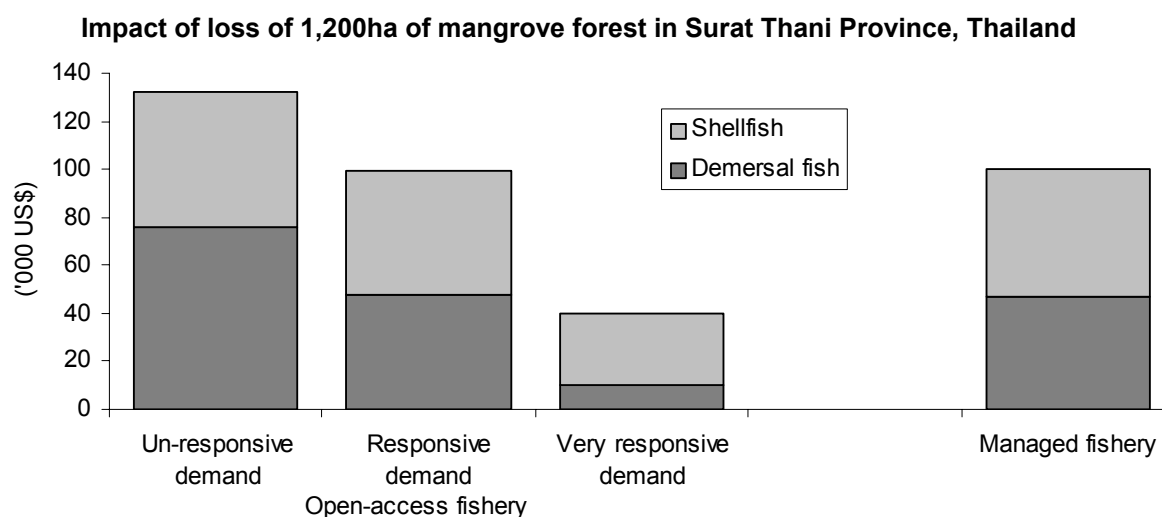
Mangrove forests can provide a number of services. These often include direct uses such as production of fuelwood and other goods, and recreation. Their most valuable services, however, are often their indirect benefits such as storm protection and their role as breeding grounds and fisheries for fish.

A large number of studies have explored the mangrove-fishery linkage. These studies generally use the production function approach: they assess the role that mangroves play as an ‘input’ into the ‘production’ of fish. This type of analysis requires two major elements. The first is an understanding of the role that mangrove forests play in the life cycle of relevant fish species. This might be arrived at either through an understanding of the biological processes at work, or by statistical analysis of the relationship between fish populations and mangrove forest condition (allowing for other factors that also contribute). The second element needed is an understanding of the markets for the products—in this case the fish. The value of mangrove forests is imputed based on how changes in their condition change the value generated in the market for the fish (holding other things constant). When there are multiple species of fish dependent on a given area of mangrove

forest, and either the biology or the markets for each species are different, these analyses would have to be conducted separately for each species.

The figure below shows the estimated consequences of loss of mangrove forest in Surat Thani Province, on the Gulf of Thailand. This region lost half its mangrove forest area in the period 1975–1993, primarily to expansion of shrimp cultivation. As can be seen, the estimated losses resulting from a loss of 1,200ha of mangrove forest (the approximate annual rate of loss in the early 1990s) depends on both the species concerned and the characteristics of the market. If the fisheries are assumed to be managed, the loss of 1,200ha of mangrove forest would cause losses of about US\$100,000. If the fisheries are assumed to be open access, the losses depend on how consumers respond to price changes: losses are highest when consumers are unresponsive (about US\$40,000), and lower when consumers are very responsive (about US\$132,000).

Note that without knowing the benefits of the land uses which replace the lost mangrove forests, we cannot conclude anything about whether society is better or worse off as a result of this deforestation.



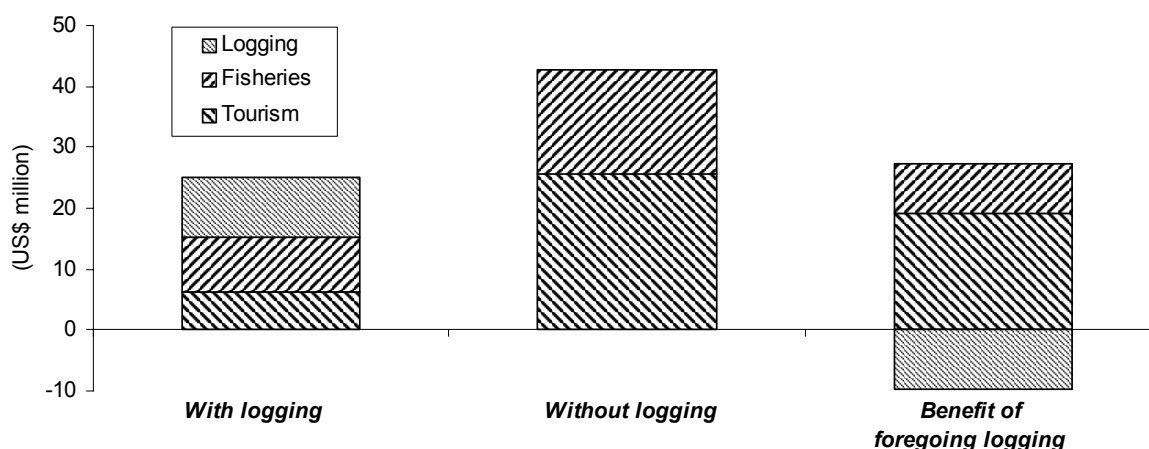
Source: Sathirathai (1997) as cited in Barbier (2000).

Case study 7: Tourism vs logging on Palawan

El Nido is a coastal town located on Bacuit Bay, in the Philippine island of Palawan. Bacuit Bay covers about 120 km² and includes 14 islands, each surrounded by fringing reefs. In 1986, marine activities in the bay included both commercial and artisanal fishing, as well as two international scuba diving resorts. Upstream logging on the land surrounding the bay, however, was having a negative impact on the bay’s water quality, threatening the viability of its fisheries and tourism industries. An analysis predicted that while logging would generate gross revenues of US\$9.8 million over 10 years (discounted at 10 percent), the increased sedimentation it was causing would result in lost revenues of US\$8.1 million from fisheries and of US\$19.3 million from tourism over the same period. Although this analysis was rather crude, for lack of better data, its results were compelling enough to lead the national government to ban logging in the Bacuit Bay watershed and declare the Bay a Marine Reserve. A more detailed analysis might have refined the numbers somewhat, but given the large gap between the benefits of logging and the costs it imposed, the overall conclusion would probably not have changed.

A resurvey of El Nido area was conducted in 1996. By then, the Bay’s coral reefs had recovered from the sedimentation damage they had suffered from logging. As the analysis had predicted, the tourism industry was flourishing—indeed, it was growing much more than predicted. This showed that preservation of the unique forest ecosystem had allowed ecotourism to flourish. Of course, new challenges have arisen: the increased growth of tourism has been accompanied by rapid growth of small businesses and guesthouses. Although these provide an alternative livelihood to local residents, unmanaged growth was becoming a threat to the ecotourism industry. Furthermore, local population expansion increased demand on fisheries resources and put severe pressure on the populations of high value marine species. Over-fishing severely reduced populations from most high valued species of fish and shellfish. Scuba divers have noticed reductions in numbers of large fish, although they are still attracted by the interesting corals, drop-offs and small reef fish. The government now faces the dilemma about how to control excessive fishing in Bacuit Bay.

Gross revenue over 10 years under alternative management options, El Nido, Philippines



Source: Hodgson and Dixon (1998), Morris (2002).

Case study 8: The costs and benefits of Madagascar's protected areas system

Stagnant agricultural yields and a growing population have led to substantial clearing of land for agricultural use in Madagascar, threatening the country's unique biodiversity. A protected areas system has been created in an effort to conserve biodiversity. These areas have succeeded in substantially slowing deforestation within their boundaries. With an estimated 70 percent of the population living below the poverty line in 2001, however, many have asked whether it makes sense to spend resources on protected areas and prevent the use of their land and timber resources.

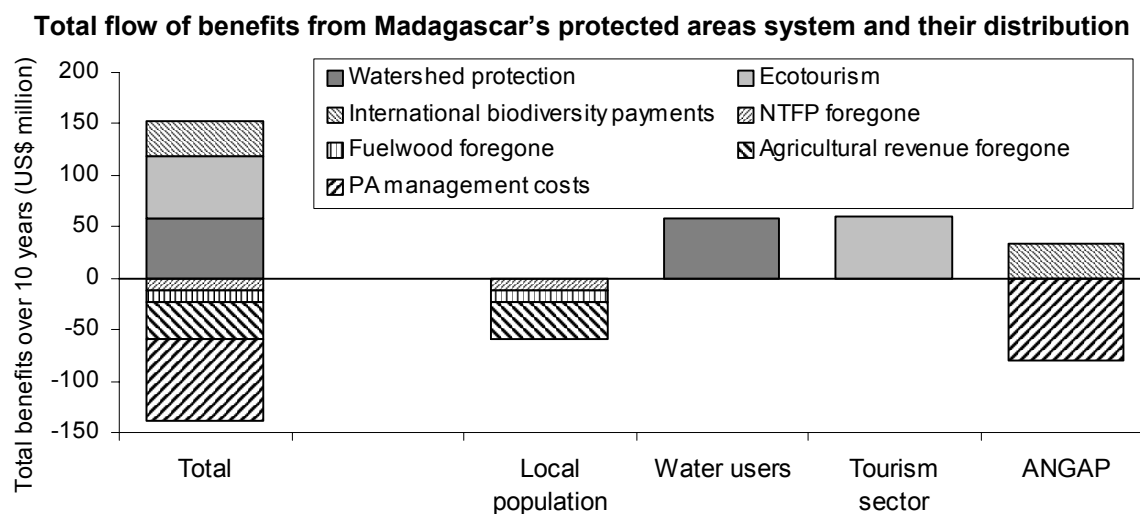
The Figure below illustrates the results of a study undertaken to estimate the costs and benefits of the protected areas system, in terms of their present value over a 10-year period. The first column shows the total flow of benefits from the protected area system. This analysis was undertaken from the country's perspective: that is, it did not include global benefits, except to the extent that the country receives payments for providing them (formally, these payments finance the costs of conservation; an avoided cost, however, is equivalent to a benefit). It also included the benefits of tourism only to the extent that they are captured by the country (although lack of data on net revenues from tourist spending limited the

analysis to entrance fees paid by visitors to protected areas).

Despite the high management costs and the foregone income from use of that land, the system is estimated to provide net benefits to the country, thanks to the valuable watershed protection services these areas provide, their tourism benefits, and the payments received for biodiversity conservation.

But as the breakdown in the right side of the figure shows, these benefits are very unevenly distributed. Local communities bear the brunt of the costs, as they are barred from using protected areas either for agriculture or for the collection of fuelwood and other NTFPs. Downstream water users such as irrigated farmers benefit substantially, as do tourism operators. The protected area management agency, ANGAP, bears the management costs but receives external support (and a part of the tourism benefits).

These results confirmed that Madagascar benefits from its protected areas system, though that depends on continued support from the global community. It also indicated the need for support to protected areas to include appropriate compensation mechanisms for local communities.



Source: Carret and Loyer (2003).

Case study 9: The impact of conservation on local communities in Uganda

Uganda’s Lake Mbuoro National Park (LMNP) covers 260 km² of open and wooded savanna and wetlands. The land and resources of the area form an important component of agro-pastoral production systems and local livelihoods. The LMNP’s establishment has significantly restricted the use that local communities—more than 50,000 people, mainly *Bairu* cultivators and *Bahima* herders—can make of the area.

Recognizing that the LMNP imposes significant opportunity costs to communities living next to it, a program of local revenue-sharing was piloted and staff were employed as community conservation officers (LMNP was the first protected area in Uganda to attempt such measures). Nevertheless, chronic funding shortages have restricted the impact of these efforts.

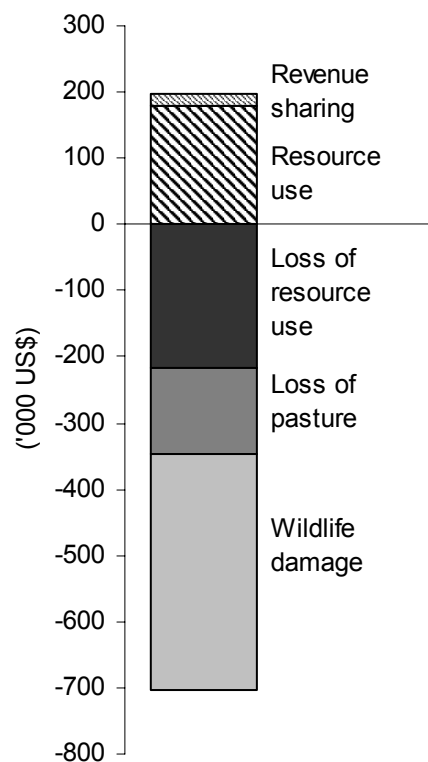
The figure below shows estimates of the costs that local communities bear because of the LMNP and the benefits they receive. The costs are estimated at about US\$700,000 a year. About half of the costs are due in damage from wildlife: more than 90 percent of households living near the park suffer regular crop destruction, livestock kills, and transmission of disease from wild animals to domestic stock. Another third of the costs are opportunity costs resulting from restrictions on product extraction in the park. The rest is due to loss of access to grazing land: the LMNP’s area includes critical dry-season grazing land sufficient to sustainably support more than 10,000 cattle and small stock. On the positive side of the ledger, local communities are able to collect products worth about US\$180,000 a year from within the park: small-scale fishing, fuelwood collection, and harvesting of other NTFPs are permitted. Local communities also receive about US\$30,000 a year through revenue-sharing arrangements.

Under these conditions, it should not be surprising that there are intense conflicts between the park and surrounding populations. Local communities are largely unwilling—and in many cases economically unable—to bear these uncompensated costs. Park authorities, already over-stretched in both budgetary and human resources terms,

continue to find it difficult to control unsustainable and illegal uses of the LMNP. Although community conservation efforts represent a major step forward in improving relations between the park authorities and nearby residents, they have to date proven inadequate to balance the high local opportunity costs of LMNP.

As in Case Study 7, it is important to note that this is a partial analysis: it only considers the costs and benefits borne by local communities. We cannot and should not conclude from these figures alone that the LMNP imposes net losses on either Uganda as a whole, or the global community. We can conclude that if either Uganda or the global community wish to continue to enjoy whatever other benefits the LMNP may be generating, then they will need to better compensate local communities for the costs they are bearing.

Costs and benefits of the Lake Mbuoro National Park for local communities



Source: Emerton (1998).

Case study 10: Paying for water services in New York State

New York City obtains its water supplies from watersheds in the Catskill Mountains, north of the City. Thanks to natural filtration, this water is of sufficient quality that it can be used unfiltered. By the end of the 1980s, however, changing agricultural practices and growing urbanization in the Catskills were threatening water quality. Non-point source pollution increased substantially, as did the threat of sewage contamination.

Threats to water quality forced city officials to consider filtering its water supply to ensure it continued to meet water quality standards. The estimated cost of a filtration facility with enough capacity and backup to process the 1.35 billion gallons a day of water that the watershed then provided the City (a successful water conservation program has since reduced this volume to about 1.1 billion gallons a day) was US\$4 to US\$6 billion dollars and the annual operating cost another US\$250 million annually, for a total of about US\$8-10 billion in present value terms. Clearly, replacing the services hitherto provided by the Catskills watershed would be expensive! Were these services to be lost, however, the City would have had little choice: building a filtration plant would have been mandated by the need to meet legal requirements for water quality.

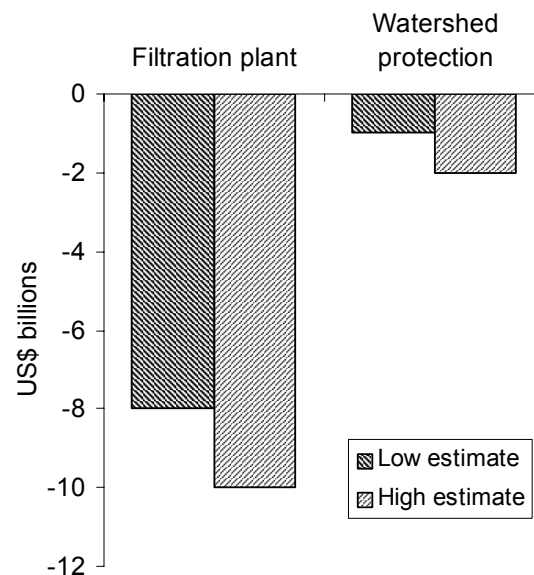
To avoid incurring this cost, the City embarked on an alternative approach: instead of paying to clean up the results of degrading the water producing environment, the City invested in preserving the rural Catskill environment that was providing it with the world's best urban water. A range of measures were adopted, the most important of which were buying particularly important areas out-right and paying farmers to operate their farms in ways which minimized water pollution. Under the latter program, known as 'Whole Farm Planning', the City pays both the operating costs of the program and the capital costs of pollution control investments on each farm. Specific pollution-control investments were designed on a farm-specific basis, with measures selected not only for their pollution control benefits, but also for their integration into the farmer's business plan, thus also bringing them significant ancillary benefits

(often in the form of time and labor savings). Within five years of the program's establishment, 93 percent of farmers in the watershed had chosen to participate. Whole Farm planning is considered to be one of the most successful non-point pollution control programs in the United States. It has played a major role in stabilizing and reducing watershed pollution loads and in enabling the City to avoid having to filter its water supply. The program to conserve the Catskills watershed cost the City about US\$1.5 billion—a considerable saving over the US\$8-10 billion that a filtration plant would have cost.

This example shows how valuation, even if only partial, can help illuminate alternative courses of action.

This example also illustrates the perils of the replacement cost technique (see Box 3.2). Clearly, the filtering wasn't the least-cost solution to the problem! Using it to value the filtration services provided by the watershed would have been a massive over-estimate.

Costs of alternative strategies to meet New York City water quality requirements



Source: NRC (2000), Heal (1999).

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