LDC and SIDS Targeted Portfolio Approach for Capacity Development & Mainstreaming of Sustainable Land Management Project

Environmental Economics Tool Kit

Analyzing the Economic Costs of Land Degradation and the Benefits of Sustainable Land Management

This document has been produced by the Global Support Unit and was written by Lars Hein, Wageningen, the Netherlands

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Preface

This Environmental Economics Toolkit is prepared in the context of the GEF 'LDC and SIDS Targeted Portfolio Approach for Capacity Development and Mainstreaming of Sustainable Land Management Project'. The Toolkit provides guidance on the application of environmental economics, and specifically the analysis and valuation of ecosystem services, to analyze the costs of land degradation and the benefits of sustainable land management.

The Toolkit is intended to assist technicians and decision makers in their analysis of land degradation and land management policy options. The Toolkit has been prepared in particular for application in LDC and SIDS countries. As much as possible, methods have been selected that have minimum data requirements, and the case studies that illustrate the methodologies reflect issues of potential relevance in LDC and SIDS countries, including coastal zone management issues.

The Toolkit contains five Tools that together present a detailed description of the various relevant ecological and economic assessment methodologies. A number of case studies illustrate the application of these methodologies. For policy makers, an executive summary is provided that describes the basic approach and its potential to support policy making in the field of land and ecosystem management.

The Toolkit is based on an in-depth literature review of (i) the theories and applications of environmental-economic valuation techniques; and (ii) the existing experiences with ecosystem services assessment in the context of sustainable land management. Furthermore, the Toolkit has benefited from comments of stakeholders involved in SLM including the Technical Advisors of the LDC and SIDS Targeted Portfolio Project.

Executive Summary

This Toolkit has been prepared to support the design and implementation of Sustainable Land Management (SLM) programs. The specific purpose of the Toolkit is: **to inform the user of the approaches that can be followed to analyze and value the economic costs of land degradation and the benefits of sustainable land management**. 'Land' is interpreted broadly in the Toolkit, also including wetlands and coastal zones.

The Toolkit follows the general approach of the Millennium Ecosystem Assessment (2003, 2005). Among others, this means that the Toolkit considers the broad range of benefits provided by agricultural and natural ecosystems, including provisioning, regulation and cultural services. The various benefits provided by land are referred to as 'ecosystem services' and they may include for example the production of food crops, the regulation of water flows, and the provision of opportunities for recreation and nature conservation. The Toolkit also specifically addresses the different scales (local, national, regional, global) at which benefits and potential costs of SLM are provided or incurred.

Ecosystem services are a central concept in this Toolkit. Economic valuation of ecosystem services can support land use policy making and implementation in various ways. First, it can reveal the economic costs and benefits of land use conversion, or of different types of land management. For instance, the economic costs and benefits of short-term exploitation of forest resources can be compared with those of sustainable management. In this way, it can also show the trade-offs in land management, i.e., the economic benefits lost and gained, and the stakeholders benefiting and losing from different policy alternatives. Second, it can show the interests of different groups of stakeholders in land and ecosystem management, thereby providing a basis for conflict resolution and integrated, participatory planning of resource management. Third, the approach allows calculation of economic efficient land management options, for instance the calculation of the optimal degree of pollution control in a lake ecosystem that is used both as waste outlet for local industries and for water supply, fishing and recreation. Fourth, it can provide the basis for setting up Payment for Ecosystem Services (PES) type of schemes, which are a market conform, innovative mechanism for allocating funds from the beneficiaries of ecosystem services to the providers of these services.

Specifically, this Toolkit contains five complementary Tools. These deal with (i) Selection of the appropriate assessment approach; (ii) Ecosystem function and services identification; (iii) Ecosystem services assessment (in bio-physical terms); (iv) Economic valuation; and (v) Ecological-economic modeling. Each Tool contains 2 or 3 subsequent steps, and a number of case studies have been added to illustrate each of the Tools.

The Toolkit allows for three types of assessments. The first type is 'Partial valuation'. This requires the application of Tools (i), (ii), (iii) and (iv) and involves the economic valuation of only one or a limited set of ecosystem services. This type of assessment can be used to show the economic benefits of a certain land use, and the costs or benefits of land use conversion with regards to specific ecosystem services. It can be applied, for instance, to assess the economic benefits of eco-tourism on a coral reef, or the economic damages resulting from a loss of wood production due to forest fires.

The second approach is 'Total valuation'. This approach also requires the application of Tools (i) to (iv), and is appropriate where a full accounting of the benefits provided by an area under a certain management system is required. In this case, all significant services need to be identified and valued. For instance, in case a decision needs to be taken involving the selection of one of two land use conversion options, it is important to analyze all benefits provided under the two options. This second approach can also be used to compare the economic benefits generated by two differently managed ecosystems, for instance an area under SLM and an area under regular management.

The third approach is the 'Impact analysis', and involves application of all 5 Tools of the Toolkit. This is a dynamic approach, which needs to be applied in case of a change in the management of a specific area. In this case, it is necessary to analyze both the economic value of the benefits generated by the system under consideration, and how the supply of these benefits will change following a change in management practices. It can be used, for instance, to analyze the economic benefits of SLM compared to traditional land management, or to assess the economic impacts of desertification.

The Toolkit describes the various Tools in detail, and explains how they can be applied to support the design and implementation of SLM programs.

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Introduction

Application of ecosystem services valuation to support SLM

There is a broad recognition that sustainable land management (SLM) is crucial for ensuring an adequate, long-term supply of food, raw materials and other services provided by the natural environment to the human society. SLM involves both the long-term maintenance of the productive capacity of agricultural lands, and the sustainable use of natural and semi-natural ecosystems, such as semi-arid rangelands or forests.

Nevertheless, SLM practices are the exception rather than the rule in many parts of the world. A whole range of social, institutional and economic factors play a role with regards to the lack of sustainability in the management of natural resources. For instance, farmers and local ecosystem users may be driven by immediate food and income requirements and may have limited possibilities to adjust harvest levels to the carrying capacity of the ecosystem. Logging companies or other external users involved in the exploitation of specific resources may disregard local interests in land and ecosystem management.

One of the factors that is often identified as being critically important is that the various economic benefits that are provided by multifunctional agricultural landscapes and natural ecosystems tend to be underestimated in decision making. Agricultural and natural ecosystems may provide a whole range of valuable goods and services, ranging from the supply of food or medicinal plants, to the regulation of water flows and biochemical cycles, to the provision of sites for recreation or cultural events. Many of these services directly or indirectly contribute to human welfare and, as such, have economic value.

The general lack of recognition of these values in decision making is caused by a range of factors. First, these benefits are often difficult to specify, as they are widely varying in terms of the type of benefit supplied, and as they operate over a range of spatial and temporal scales. Second, several of these benefits have a public goods character and/or are not traded in a market. In spite of their welfare implications, they therefore do not show up in economic statistics. Third, there is often a mismatch between the stakeholders that pay the (opportunity) costs of maintaining an environmental benefit (e.g. by not converting a forest to cropland) and the beneficiaries of that benefit (e.g. downstream water users benefiting from the regulation of water flows).

Through assessment of the economic value of the multiple benefits provided by land and ecosystems, it is possible to increase the awareness of stakeholders and decision makers of the economic benefits resulting from sustainable land management. Since economic considerations generally play a key role in decision making, it is anticipated that economic valuation of environmental benefits can contribute to a more sustainable and a more efficient decision making. Analysis and valuation of ecosystem services can also guide the setting up of mechanisms to compensate the suppliers of ecosystem services for the costs related to providing those benefits in a Payment for Ecosystem Services (PES) mechanism.

However, the economic value attached to environmental resources should always be seen as covering only one of a set of decision making criteria. It deals only with the economic (or efficiency) impacts of decision making, and does not yield any information on, for instance, equity issues. It is also deficient in that not all values can always be meaningfully transformed into an economic value estimate (such as the value of a protected species). These various constraints to economic valuation of environmental benefits are elaborated in the Toolkit (in Tool 4 'Economic Valuation'), and guidance is provided on the precise scope and potential contribution of economic valuation for the promotion of SLM practices.

This Toolkit provides guidance on the use of environmental economics, and in particular ecosystem services valuation, to support the design and implementation of SLM programs and

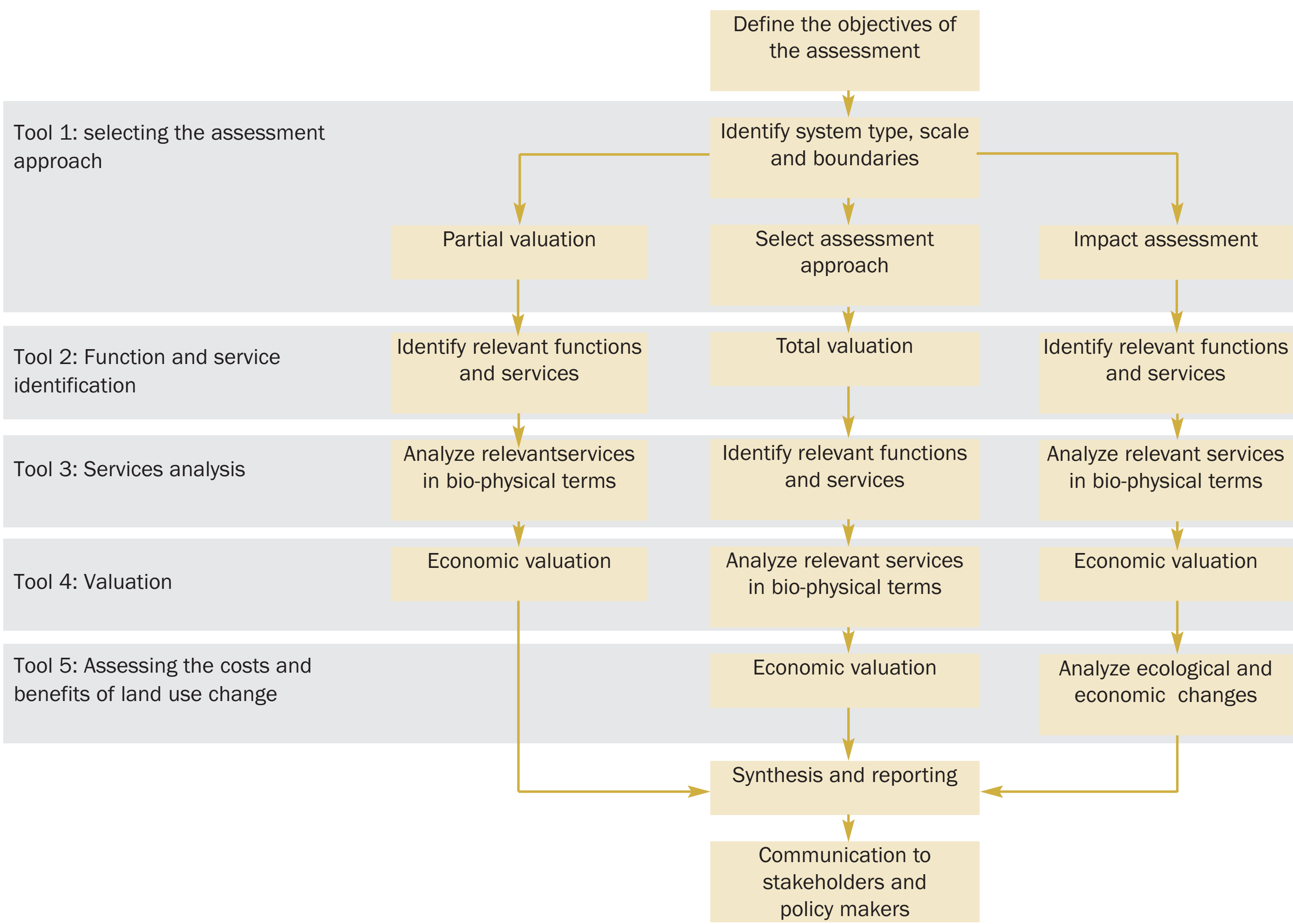

Figure 1. Structure of the Environmental Economics Toolkit

investments. The general approach of the Toolkit is in line with the Millennium Ecosystem Assessment (2003). The specific purpose of the Toolkit is to enable the user to analyze and value the economic costs of land degradation, and the benefits of sustainable land management. The Toolkit includes several case studies that illustrate the described methodologies, as well as suggestions for further reading.

Structure of the Toolkit

The Toolkit comprises five complementary tools: (i) Selection of the appropriate assessment approach; (ii) Identification of ecosystem functions and services; (iii) Bio-physical assessment of ecosystem services; (iv) Economic valuation of ecosystem services; and (v) Ecological-economic modeling. The Toolkit can be used for three distinct approaches to analyzing the economic benefits of SLM: Partial Valuation, Total Valuation and Impact Assessment, see Figure 1.

Partial valuation involves the economic analysis of only one or a limited set of services derived from an ecosystem. Total Valuation is more comprehensive than Partial Valuation, involving the analysis of all significant ecosystem services. It is more accurate, but also more data intensive than the previous method. Impact Assessment requires an additional step, involving the analysis of how changes in land use or management will influence ecosystem services supply. This approach is more data intensive than the previous two, and requires additional analysis related to the modeling of the dynamics of the ecosystem. Table 1 explains the various ways in which the three types of assessment can support policy design and implementation.

In addition to the applications described in Table 1, the valuation of ecosystem services may also support the formulation of Payment for Ecosystem Services (PES) schemes. These schemes involve the monitoring of ecosystem services, and the subsequent allocation of payments from beneficiaries to suppliers of ecosystem services. These schemes can support SLM in case there are large discrepancies between the stakeholders benefiting from SLM, and the stakeholders responsible for management of the ecosystem, for instance in the case of downstream water users and upstream stakeholders responsible for the maintenance of upland forest that regulate the downstream water flows (see Box 1).

Application of the Toolkit requires a multidisciplinary approach. Depending on the area and ecosystem involved, the analysis may require ecological, hydrological, soil sciences, spatial modeling, policy sciences, anthropology and economic inputs, and the assessing team needs to cover the range of relevant disciplines. In most cases, this includes at least ecology and economics.

The Toolkit will not elaborate on the communication of results to stakeholders and policy makers, as this is the topic of several other programs in the development field, e.g. in the context of the GEF/UNDP/UNEP National Communications Support Program. Note, however, that the Toolkit can support communications to decision makers at different stages of the decision making process. First, identification of services and stakeholders (Tool 2) may guide consultative processes to be undertaken as part of a decision making process, by revealing the stakeholders with an interest in the land management issue at stake. Second, the Toolkit can be applied to inform decision makers of the economic implications of potential land use change and land use policy options. Third, combined with an optimization study (Tool 5, Step 5.3), the Toolkit can advise policy makers with regards to optimal responses to environmental issues.

Table 1

Three approaches to analyze the costs of land degradation and the benefits of SLM.

Approach	Potential to support cost-benefit analysis of land degradation and SLM
Partial Valuation	This approach is useful where only few services provide the majority of the benefits to society, or where analysis of only few benefits is required to support decision making. In terms of assessing land degradation, it can be used to compare the key benefits provided by a degraded and a non-degraded system, or a sustainably and a non-sustainably managed ecosystem. Provided that the ecosystems are otherwise comparable (in terms of ecosystem type, socio-economic environment, etc.), this comparison will indicate the overall costs of land degradation and the benefits of SLM.
Total Valuation	Total valuation is appropriate where a full accounting of the benefits provided by an area under a certain management system is required. This approach can be used, for instance, in case the benefits of two land use conversion options need to be compared. It can also be used to compare the benefits of two differently managed ecosystems, for instance an ecosystem under SLM and an ecosystem under regular management.
Impact Assessment	This approach can be used to analyze the costs of a continuous, progressive degradation of an ecosystem, or the benefits of applying SLM in a specific ecosystem. For instance, in case a rangeland manager decides to adopt a sustainable rangeland management package (involving e.g. rotational grazing, fire control, seeding of enhanced grasses, optimal stocking, etc.), this will gradually change the species composition and productivity of the rangeland. Impact Assessment is required to understand the changes in the ecosystem, and to subsequently analyze the economic benefits of the new management regime.

BOX 1. Payment for Ecosystem Services (PES) Schemes

In recent years, PES schemes have emerged as an innovative option to provide incentives for sustainable ecosystem management. PES schemes require the valuation of selected ecosystem services, the identification of beneficiaries and providers of the services, and the set-up of a payment scheme that regulates the transfer of payments from beneficiaries to providers in return for maintaining the supply of the ecosystem service. PES approaches have been applied in a range of settings. For instance, the U.S. government spends over US$1.7 billion per year to induce farmers to protect land. In Latin America, particularly Costa Rica and Mexico, various stakeholders such as irrigation water-user groups, municipal water supply agencies and other governmental bodies have initiated and executed PES schemes aimed at maintaining downstream water supply. Other examples are provided by Conservation International, which is protecting 81,000 hectares of rainforest in Guyana through a conservation concession that costs US$1.25 per hectare per year, and the Wildlife Foundation in Kenya, which is securing migration corridors on private land through conservation leases at US$ 4 per acre per year (UNEP, 2005). The major benefit of PES schemes is that they can provide a long-term flow of funds necessary to protect certain ecosystem services. However, care needs to be taken in the set-up of new PES schemes. Transaction costs can be very high, both with respect to setting up the PES scheme including a trustworthy fund manager, and for monitoring the flows of ecosystem services that provide the basis for the payments. In addition, PES schemes are unlikely to be successful if local beneficiaries are poor and have no funds available to pay for the ecosystem services they receive.

Application of the Toolkit; an illustration

In order to further explain how the Toolkit can be applied, an illustration of each Tool is presented, based on a hypothetical case study in which the costs of land degradation are analyzed (see Table 2).

Table 2.

Analyzing the costs of land degradation; an example.

Step	Purpose	Illustrative example
1. Problem definition	Define the study area, the type of land degradation involved, the relevant temporal and spatial scales and potentially relevant institutional aspects, e.g. land tenure.	1. The hypothetical study area is an African subhumid ecosystem, where maize, cowpea, millet and cotton are grown in a varying landscape consisting of a river bordered by a plain and surrounding hillsides. 2. The two key types of land degradation are soil nutrient depletion and erosion. 3.The study deals with the local and national impacts of land degradation, and has a time horizon of 20 years.

Step	Purpose	Illustrative example
2. Ecosystem functions and services identification	To identify the key functions and services provided by the system	The system supplies the following services: 1. Food production through irrigated, non-irrigated lowland, and non-irrigated upland agriculture; 2. Grazing and animal production; 3. Provision of hunting opportunities on fallow lands; 4. Control of erosion and sedimentation rates by vegetation in uplands.
3. Ecosystem services assessment	Quantification of the services in biophysical terms	1. The system supplies x ton of maize, y ton of millet, and z ton of cotton, requiring a units of fertilizers, seeds, equipment and labor; 2. Off-take rate: # of animals slaughtered per year, at b labor costs and c other costs (fencing, veterinary services); 3. Fallow lands provide x ton of bushmeat per year, at labor costs d; 4. Erosion rates in the river are controlled by upland vegetation. With vegetation, only o ton of sediments would be deposited in the river, without vegetation this would increase to p ton. Loss of vegetation would increase sedimentation of a downstream hydropower dam with q tons of sediments per year.
4. Economic valuation	Expressing the services in a monetary value	1. Net value of US$ v per year generated by irrigated and upland and lowland non-irrigated agriculture; 2. The monetary value of meat, skins and milk is US$ w per year; 3. The monetary value of bushmeat is US$ x per year; 4. The costs of sedimentation in the dam amount to US$ e per year.
5. Assessing the costs and benefits of land use change	Analyzing the impacts of changes in the landscape on ecosystem services supply	1. Soil nutrient depletion is lowering crop yields in upland and lowland crops with a ton per year, causing an economic loss of US$ u per year; 2. Erosion is leading to loss of upland crops of b ton per year, causing an economic loss of US$ y per year; 3. Reduced fallow periods, loss of vegetation cover, and high hunting pressures have reduced bushmeat harvest with c ton per year, causing an economic loss of US$ w per year; 4. Siltation of downstream sediments is reducing the lifetime of the reservoir with d years, causing economic losses of US$ x.

TOOL 1 Selecting the assessment approach

Introduction

The first step in the application of the Environmental Economics Toolkit is to determine the overall objective or problem to be analyzed. As indicated in Figure 1, the type of problem will determine the overall economic assessment approach that needs to be followed. Examples of the type of analysis that can be conducted with the Environmental Economics toolkit are (i) the analysis and valuation of the benefits of adopting SLM practices; (ii) analysis of the economic costs of land degradation; or (iii) comparison of two project alternatives with different environmental and land management impacts.

Once the objective or problem is clearly defined, the user needs to select the appropriate assessment approach and the units of the analysis. The Toolkit allows for three types of Assessments: Partial Valuation, Total Valuation and Impact Assessment. The object of the study can either be an ecologically defined system, such as a forest plot or a watershed, or an institutionally defined system, such as a municipality or a country. The area can be relatively homogeneous, including only one main ecosystem type (e.g. a semi-arid rangeland), or it can be heterogeneous (e.g. comprising a mix of agricultural and semi-natural lands). In case the area comprises different systems, it is likely that the sub-systems supply different types of ecosystem services, which needs to be accounted for in the application of Tools 2 and 3.

Subsequently, the user needs to specify the system boundaries including the relevant spatial and temporal scales for the assessment. The benefits of an area may accrue to stakeholders at different scales, ranging from local farmers or users, to regional traders, to national investors, to the global community that, for example, may have an interest in globally important biodiversity contained in a system. The user needs to decide if the assessment will extend to all stakeholders, or if it will be confined to particular scales (e.g. the impact of SLM on local food security; or the global costs of land degradation). It is also important to select the appropriate time horizon for the assessment: is the objective of the assessment to determine the current flows of benefits, or is the long-term supply of benefits relevant ?

Purpose of the Tool

The purpose of the first tool is to guide the user in clearly defining the objective of the economic assessment and the system to be studied, and to assist the user in selecting the appropriate valuation approach.

How to use the Tool

The tool provides the starting point for analyzing ecosystem services in the context of sustainable land management. It contains three steps, dealing with: (i) problem definition; (ii) selecting the unit of analysis; and (iii) specification of systems boundaries.

Step 1.1 Problem definition and selection of the valuation approach

The first step to be carried out in this Tool is the selection of the appropriate assessment procedure. As explained in the Introduction section, the user may be interested in (i) Partial valuation; (ii) Total valuation; or (iii) Impact assessment. Some examples of potential applications of the 3 valuation types are provided in Table 3 below.

(i) Partial valuation. Partial valuation involves the economic valuation of only one or a limited set of environmental benefits. It can be used where only few environmental benefits supply the large majority of benefits to society, and where appraisal of only few benefits is required to support decision making. This approach can be applied, for instance, in case the impact of SLM on food security needs to be assessed.

(ii) Total valuation. The second approach is 'Total valuation'. This approach is appropriate where a full accounting of the benefits provided by an area under a certain management system is required. In this case, all services need to be identified and valued. For instance, in case a decision needs to be taken involving the selection of one of two land use conversion options, it may be important to analyze all benefits provided in the two options. Note that, in specific cases, it may be clear that some services only generate a very minor part of the total benefits, as in the case of carbon sequestration in a system that absorbs only minimal amounts of carbon over time. In this case, it may be decided to skip these minor services and include them only as a pro memory post.

(iii) Impact assessment. The third approach is 'Impact assessment'. It involves analyzing the impacts of changes in environment and land management on the supply of benefits

Table 3.

Examples of the potential applications of the three main valuation approaches

Valuation approach	Examples
Partial valuation	1.Valuation of the productive capacity of a semi-arid rangeland. 2.Valuation of the production of wood and/or Non-timber forest products from a specific forest, or the forests in a country. 3.Valuation of the hydrological service of an upland forest in order to define a payment vehicle from downstream users to upland managers to maintain this service.
Total valuation	1.Valuation of the ecosystem services supplied by a forest in order to compare the benefits of timber logging with those of sustainable management 2.Valuation of the services provided by a natural area in order to identify which stakeholders benefits from the area and which stakeholders may be expected to contribute to financing the preservation of the area.
Impact assessment	1.Analyzing the impacts of pollution control measures in a wetland on water quality and ecosystem services supply in order to compare the costs and benefits of pollution control measures 2.Analysis of the impact of disturbances (e.g. road construction, or desertification) on the supply of ecosystem services.

to society. This approach needs to be applied in case of a change in the management of an area (e.g. through the adoption of various SLM practices). In this case, it is necessary to analyze both the economic value of the benefits generated by the system under consideration, and how the supply of these benefits will change following a change in management practices. This approach is also relevant for the prediction of the impact of environmental pressures, e.g. pollution, that may cause a change in the state of the environmental system. Hence, compared to the two previous approaches, this approach requires an additional Tool, dealing with how the impact of the change in management or pressures can be analyzed or modeled.

Step 1.2 Defining the unit of analysis

A key question that every user of these guidelines will come across at some stage is 'should the services supplied by this ecosystem be valued in monetary terms or not ?', to be followed by 'should all or only some services be valued in monetary terms, and what do we do with the other services ?'

It is clear that monetary valuation is no 'silver bullet' that provides a unequivocal approach to measure the full value of world's ecosystems. Besides the practical problems that may occur in measuring services, it is fundamentally difficult to translate subjective values dealing with health, peoples lives, and nature into the single unit 'money'. People do not normally express everything along one value type, but are used to thinking of multiple value types (see e.g. Martinez-Alier et al. 1998; O'Neill, 2001 and Munda, 2004 for more information).

However, on the other side, where decisions are made in formal fora, decision makers require some kind of a unit with which to compare costs and benefits of different policy options, and the most commonly used unit is a monetary one. Hence, there is a need to make sure that as much as reasonably possible, ecosystem services are expressed in a monetary unit in order to be properly accounted for in decision making processes. Therefore, in these guidelines, the practical recommendation is to express as much services in a monetary unit as is possible from a theoretical and a practical perspective. For all production and most regulation services, it will, in principle, be possible to estimate the monetary value of the service, as most of these services can be either directly or indirectly related to a market transaction. In the case of the cultural services, this is much more complex. For instance, it may often not be possible to translate the full value of biodiversity and nature in a monetary unit, as the economic value of a species, or a population of a species, is in most cases very hard to determine.

In view of the above, money will be the unit of choice for these guidelines, complemented with specific indicators for those services that are hard to express in monetary terms (such as biodiversity, as further specified in Tool 3 ('Bio-physical Assessment'). The monetary value of an ecosystem service can be expressed either (i) in terms of an annual value indicating the flows of benefits form an ecosystem (e.g. US$/ha/year); or (ii) as Net Present Value (NPV), which indicates the sum of the present and discounted future flows of net benefits from the ecosystem (e.g. US$/ha). In this second case, future flows are discounted with a discount rate in order to account for the preference people have for money now rather than at a later stage. The concept of NPV is further elaborated in Tool 4.

Step 1.3 Defining the system boundaries

Valuation (as any other analysis) requires that the object of the valuation is clearly defined. Hence, it is necessary to define the system to be analyzed, in terms of its spatial and temporal boundaries. The ecosystem is the entry point often used for valuation of ecosystem services and environmental benefits.

The Convention on Biological Diversity provided the following definition of an ecosystem "a dynamic complex of plant, animal and micro-organism communities and their nonliving environment interacting as a functional unit" (United Nations, 1992). For the purpose of this Toolkit, this definition is further operationalised following Likens (1992) who elaborates on the spatial aspects of ecosystems: 'Ecosystems are the individuals, species and populations in a *spatially defined area*, the interactions among them, and those between the organisms and the abiotic environment'. This spatial approach makes it easier to define the physical boundaries of the area to be analyzed. Following the Millennium Ecosystem Assessment, ecosystems may comprise both natural and/or strongly man influenced systems such as agricultural fields.

Note that the ecosystem to be valued may contain a number of different (sub-)ecosystems. For instance, a forest ecosystem may contain open patches or a set of lakes or ponds. Spatial heterogeneity is the rule rather than the exception, and the user of the guidelines needs to be aware that ecological sub-systems may supply entirely different ecosystem services than the overall study area. Hence, a choice needs to be made in terms of system boundaries: are fundamentally different ecological sub-systems to be included in the analysis or not ? For instance, are the ponds present in a forest to be included in the analysis or not ? The response will entirely depend on the formulated problem. For instance, if the user is interested only in wood and non-timber forest products (as in a partial valuation), he may prefer to exclude the ponds from the analysis. If, on the other hand, full valuation including biodiversity aspects is the study objective, the ponds need to be included because they will have a different species composition and because they may be essential for supporting biodiversity in the forest at large.

Note that ecological and institutional boundaries seldom coincide, and that stakeholders in ecosystem services often cut across a range of institutional zones and scales. In other words, the ecosystem may be located in different municipalities or even countries. Whereas for the analysis of land degradation processes, ecosystem services and ecosystem dynamics the ecosystem is the appropriate unit of analysis, in the identification of policy measures the administrative and institutional contexts need to be explicitly considered. This incongruence between ecological and political boundaries is very common in environmental management, and flexible solutions need to be identified on a case-by-case basis. In case the benefits and costs of SLM accrue to different countries, e.g. where an upper watershed is protecting downstream river flows in another country, economic analysis of costs and benefits could be used to support PES schemes in order to compensate the first country for the supply of the ecosystem services (see Box 1).

Furthermore, the temporal boundaries of the system to be analyzed have to be defined. For partial and total valuation, the user of the guidelines may be interested in a 'snapshot' analysis of the benefits supplied by the ecosystem. In this case, the NPV would be based on the assumption that the future flows of ecosystem services would be equal to the present flows. This, clearly, is a very strong assumption, and there are numerous ecosystems (think of many fish stocks or forests) where unsustainable harvest rates are being applied, and where future flows of ecosystem services can only be expected to decline. In case of unsustainably managed ecosystems, a snap-shot analysis based on current flows of ecosystem services could severely overestimate the value of the ecosystem under current management. Hence, a snap-shot approach is only valid in case there are grounds to assume that the extraction of ecosystem services do not exceed the regenerative capacity of the ecosystem. Otherwise, reductions in flows have to be accounted for, and a longer time horizon needs to be accounted for. In this case, as well as in the case of the dynamic Impact Assessment, it is up to the user to chose the time horizon for the analysis, which can vary from for instance 20 to 50 years depending on the time frame relevant for the user, and the discount rate used.

The Convention on Biological Diversity provided the following definition of an ecosystem "a dynamic complex of plant, animal and micro-organism communities and their nonliving environment interacting as a functional unit" (United Nations, 1992).

TOOL 2 Ecosystem Function & Services Identification

Introduction

In the early 1970s, the concept of ecosystem function was proposed to facilitate the analysis of the benefits that ecosystems provide to society. An ecosystem function can be defined as "the capacity of the ecosystem to provide goods and services that satisfy human needs, directly or indirectly". Ecosystem functions depend upon the state and the functioning of the ecosystem. For instance, the function 'production of firewood' is based on a range of ecological processes involving the growth of plants and trees that use solar energy to convert water, plant nutrients and CO_2 to biomass.

A function may result in the supply of ecosystem services, depending on the demand for the good or service involved. Ecosystem services are the goods or services provided by the ecosystem to society (following the definition of the Millennium Ecosystem Assessment, 2003). The supply of ecosystem services will often be variable over time, and both actual and potential future supplies of services should be included in the assessment. Ecosystem functions, and the services attached to these functions, vary widely as a function of the type of ecosystem and the socio-economic setting involved. For example, the capacity of the ecosystem to provide firewood depends on the forest cover and the amount of woody plant biomass contained in the system, as well as, in the longer term, on the primary productivity of the forest. However, the actual supply of firewood also depends on the demand of different stakeholders for firewood. This demand is determined by the need for wood energy as well as the availability of other sources to satisfy household energy needs.

Hence, identification of functions and supplied ecosystem services is the first step in analyzing the benefits provided by an ecosystem to society. In itself, it allows a qualitative analysis of the potential consequences of environmental change, and it also provides the basis for the next steps of the Toolbox. Specification of the functions and services to be studied is also required to avoid double-counting of benefits, which may lead to overestimation of the economic benefits of an area. Furthermore, analysis of the different stakeholders that benefit from ecosystem services can assist in determining stakeholder interests in the management of an area or ecosystem.

Purpose of the Tool

The ecosystem function and services identification tool allows the user to identify, from a detailed listing, the ecosystem services relevant for the environmental and socio-economic setting under consideration. It also facilitates analyzing stakeholder interests in the management of an area.

How to use the tool

The user is recommended to first identify the functions and services relevant for his analysis, using Table 2. These relevant functions and services will depend on the objective of the analysis to be undertaken as well as the area under consideration. Second, the user should consider the issue of double counting, i.e. remove services from the list that would lead to inconsistencies in the value estimates because of the double counting of services. Third, the user is recommended to identify the relevant stakeholders and scales for each of the selected services. This will guide the further analysis and valuation of the services. These three steps are described below.

Step 2.1 Identification of functions and services

The analysis of ecosystem services for SLM starts with the identification of the functions, and the services provided by the ecosystem under consideration. In case of a 'Total valuation', all potentially relevant services need to be considered, whereas in case of a 'Partial valuation', a selection can be made based on the purpose of the assessment (as described in Tool 1). An 'Impact assessment' can be based on either valuation of all services, or of a selection.

Table 4 provides a comprehensive list of ecosystem services, containing 24 different types of services. By and large, the list follows the MEA (2003). Compared to MEA (2003) some minor adjustments have been made in order to ensure consistency in its application to SLM. The list contains three types of ecosystem services, which are based on a different type of interaction between people and ecosystems. The three types of functions are:

(i) Provisioning Services. Provisioning services are the goods and services produced by or in the ecosystem, for example a piece of fruit or a plant with pharmaceutical properties. The goods and services may be provided by natural, semi-natural and agricultural systems and, in the calculation of the value of the service, the relevant production and harvest costs have to be considered.

(ii) Regulation services. Regulation services result from the capacity of ecosystems to regulate climate, hydrological and bio-chemical cycles, earth surface processes, and a variety of biological processes. These services often have an important spatial aspect; e.g. the flood control service of an upper watershed forest is only relevant in the flood zone downstream of the forest. The nursery service is classified as a regulation service. It reflects that some ecosystems provide a particularly suitable location for reproduction and involves a regulating impact of an ecosystem on the populations of other ecosystems.

(iii) Cultural services. They relate to the benefits people obtain from ecosystems through recreation, cognitive development, relaxation, and spiritual reflection. This may involve actual visits to the area, indirectly enjoying the ecosystem (e.g. through nature movies), or gaining satisfaction from the knowledge that an ecosystem containing important biodiversity or cultural monuments will be preserved. The latter may occur without having the intention of ever visiting the area (Aldred, 1994). The cultural services category also includes the habitat service, that represents the benefits that people obtain from the existence of biodiversity and nature (not because biodiversity provides a number of services, but because it is important in itself). In this way, the list deviates from the MEA, 2003, where biodiversity is assumed to support the supply of other services by enhancing ecosystem functioning and resilience, but where the value of biodiversity in itself is not explicitly recognized. However, this does not do justice to the importance of protecting biodiversity in natural parks for the purpose of conserving biodiversity in itself. Therefore, the habitat service is added to the list. Because the importance attached to biodiversity is strongly dependent on the cultural background of the

Table 4.

List of ecosystem services (based on Ehrlich and Ehrlich, 1981; Costanza et al., 1997; DeGroot et al., 2002; Millennium Ecosystem Assessment, 2003; Hein et al., 2006).

Category		Ecosystem services
Provisioning services	Provisioning services reflect goods and services extracted from the ecosystem	• Food • Fodder (including grass from pastures) • Fuel (including wood and dung) • Timber, fibers and other raw materials • Biochemical and medicinal resources • Genetic resources • Ornamentals
Regulation services	Regulation services result from the capacity of ecosystems to regulate climate, hydrological and bio-chemical cycles, earth surface processes, and a variety of biological processes	• Carbon sequestration • Climate regulation through control of albedo, temperature and rainfall patterns • Hydrological service: regulation of the timing and volume of river flows • Protection against floods by coastal or riparian systems • Control of erosion and sedimentation • Nursery service: regulation of species reproduction • Breakdown of excess nutrients and pollution • Pollination • Regulation of pests and pathogens • Protection against storms • Protection against noise and dust • Biological nitrogen fixation (BNF)
Cultural services	Cultural services relate to the benefits people obtain from ecosystems through recreation, cognitive development, relaxation, and spiritual reflection	• Provision of cultural, historical and religious heritage (e.g. a historical landscape or a sacred forests) • Scientific and educational information • Opportunities for recreation and tourism • Amenity service: provision of attractive housing and living conditions • Habitat service: provision of a habitat for wild plant and animal species

observer, the service is classified as a cultural service (cf Hein et al., 2006).

Furthermore, contrary to Millennium Ecosystem Assessment (2003), but analogous to Costanza et al. (1997) and Hein et al. (2006), there is no category 'supporting services'. Supporting services represent the ecological processes that underlie the functioning of the ecosystem. Their inclusion in valuation may lead to double counting as their value is reflected in the other three types of services. In addition, there are a very large number of ecological processes that underlie the functioning of ecosystems, and it is unclear on which basis supporting services should be included in, or excluded from a valuation study. Therefore, in order to ensure maximum practical applicability of the Toolkit, this category of services is not further considered here.

Note that, in principle, the user has the choice of valuing services or functions; both express the benefits supplied by the natural environment to society. The main difference is that valuation of services is based on valuation of the flow of benefits, and valuation of functions is based on the environment's capacity to supply benefits. The first expresses clearly the current benefits received, but additional analyses are required if the flow of ecosystem services is likely to change in the short or medium term (e.g. if current extraction rates are above the regenerative capacity of the ecosystem). In this case, calculation of the NPV requires that assumptions are made on the future flows of services.

Functions better indicate the value that can be extracted in the long-term, and their value is not biased by temporary overexploitation. However, it is often much more difficult to assess the capacity to supply a service than to assess the supply of the service itself. For instance, for the function 'supply of fish', this requires analysis of the sustainable harvest levels of the fish stocks involved which needs to be based on a population model including reproduction, feed availability and predation levels. Hence, in most valuation studies, it is chosen to value services rather than functions, and to account for potential changes in services supply in the assessment.

Step 2.2 Screen the list for potential double counting of services

An important issue in the valuation of ecosystem services is the double counting of services (Millennium Ecosystem Assessment, 2003; Turner et al., 2003). Specifically, there is a risk of double counting in relation to the regulation services that support the supply of other services from an ecosystem. For example, consider a natural ecosystem that harbors various populations of pollinating insects. These insects pollinate both the plants inside the natural ecosystem, and the fruit trees of adjacent orchards. In an analysis of the economic value of the natural area, only the pollination of the adjacent fruit trees should be included as a regulation service. As for the various trees inside the natural area, the produce from these trees (e.g. wood, rattan and fruits) should be included in the valuation (as provisioning services), but the pollination of these natural trees should not, as this would lead to double counting

In general, regulation services should only be included in the valuation if (i) they have an impact outside the ecosystem to be valued; and/or (ii) if they provide a direct benefit to people living in the area (i.e. not through sustaining or improving another service). The first case is illustrated by the example of the fruit trees above. An example of a service that may provide a direct benefit inside an area that is not included in other ecological services, is the service 'protection against noise and dust' provided by a green belt besides a highway. If this affects the living conditions of people living inside the study area, it needs to be included in the valuation. A prerequisite for applying this approach to the valuation of regulation services is that the ecosystem is defined in terms of it's spatial boundaries – otherwise the external impacts of the regulation services can not be precisely defined.

Step 2.3 Identification of relevant scales and stakeholders

Ecological and institutional scales. Scales refer to the physical dimension, in space or time, of phenomena or observations (O'Neill and King, 1998). According to its original definition, ecosystems can be defined at a wide range of spatial scales (Tansley, 1935). These range from the level of a small lake up to the boreal forest ecosystem spanning several thousands of kilometers. As it is usually required to define the scale of a particular analysis, it has become common practice to distinguish a range of spatially defined ecological scales (Holling, 1992; Levin, 1992). They vary from the level of the individual plant, via ecosystems and landscapes, to the global system - see Figure 2.

Ecosystem services are generated at all ecological scales. For instance, fish may be supplied by a small pond, or may be harvested in the Pacific Ocean. Biological nitrogen fixation enhances soil fertility at the ecological scale of the plant, whereas carbon sequestration influences the climate at the global scale.

In the socio-economic system, a hierarchy of institutions can be distinguished (Becker and Ostrom, 1995; O'Riordan et al., 1998). They reflect the different levels at which decisions on the utilization of capital, labor and natural resources are taken. At the lowest institutional level, this includes individuals and households. At higher institutional scales can be distinguished: the communal or municipal, state or provincial, national, and international level (see Figure 2). Many economic processes, such as income creation, trade, and changes in market conditions can be more readily observed at one or more of these institutional scales.

Scales of ecosystem services. The ecological and institutional scales of ecosystem services are elaborated for each category of ecosystem services.

Provisioning services. The possibility to harvest products from natural or semi-natural ecosystems depends upon the availability of the resource, or the stock of the product involved. The development of the stock is determined by the development of the ecosystem as a function of ecological processes and human interventions. To analyze the ecological impacts of the resource use, or the harvest levels that can be (sustainably) supported, the appropriate scale of analysis is the level of the ecosystem supplying the service (e.g. the lake, or the Northern Atlantic ocean) (Levin, 1992). The benefits of the resource may accumulate to stakeholders at a range of institutional scales (Turner et al.,

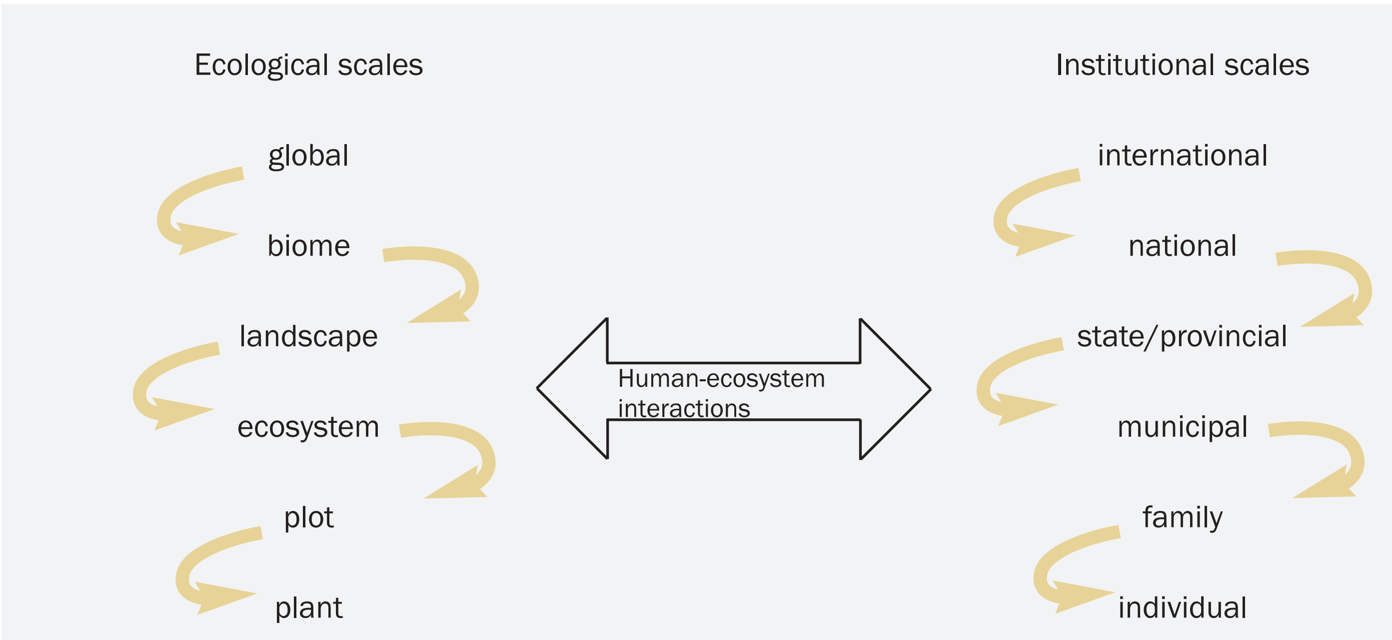

Figure 2. Selected ecological and institutional scales

2000). Local residents, if present, are often an important actor in the harvest of the resources involved, unless they do not have an interest in, or access to the resource (e.g. due to a lack of technology, or because the ownership or user-right of the resource resides with other stakeholders). In addition, there may be stakeholders' interests at larger scales if the goods involved are harvested, processed or consumed at larger scales. For example, this is the case if a marine ecosystem is fished by an international fleet, or if a particular genetic material or medicinal plants is processed and/or consumed at a larger institutional scale (see e.g. Blum, 1993).

Regulation services. A regulation service can be interpreted as an ecological process that has (actual or potential) economic value because it has an economic impact outside the studied ecosystem and/or if it provides a direct benefit to people living in the area (see the previous section). Because the ecological processes involved take place at certain, ecological scales, it is often possible to define the specific ecological scale at which the regulation service is generated (see Table 5). For many regulation services, not only the scale, but also the position in the landscape plays a role – for example, the impact of the water buffering capacity of forests will be noticed only downstream in the same catchment (Bosch and Hewitt, 1982). Stakeholders in a regulation service are all people residing in or otherwise depending upon the area affected by the service.

Cultural services. Cultural services may also be supplied by ecosystems at different ecological scales, such as a monumental tree or a natural park. Stakeholders in cultural services can vary from the individual to the global scale. For local residents, an important cultural service is commonly the enhancement of the aesthetic, cultural, natural, and recreational quality of their living environment. In addition, in particular for indigenous people,

Table 5.

Most relevant ecological scales for the regulation services – note that some services may be relevant at more than one scale. Based upon Hufschmidt et al. (1983), Kramer et al. (1995); Van Beukering et al. (2003); Hein et al. (2006).

Global	> 1,000,000 km2	Carbon sequestration Climate regulation through regulation of albedo, temperature and rainfall patterns
Biome – landscape	10,000-1000,000 km2	Regulation of the timing and volume of river and ground water flows Protection against floods by coastal or riparian ecosystems Regulation of erosion and sedimentation Regulation of species reproduction (nursery service)
Ecosystem	1-10,000 km2	Breakdown of excess nutrients and pollution Pollination (for most plants) Regulation of pests and pathogens Protection against storms
Plot – plant	< 1 km2	Protection against noise and dust Control of run-off Biological nitrogen fixation (BNF)

ecosystems may also be a place of rituals and a point of reference in cultural narratives (Posey, 1999; Infield, 2001). Nature tourism has become a major cultural service in Western countries, and it is progressively gaining importance in developing countries as well. Because the value attached to the cultural services depends on the cultural background of the stakeholders involved, there may be very different perceptions of the value of cultural services among stakeholders at different scales. Local stakeholders may attach particular value to local heritage cultural or amenity services, whereas national and/or global stakeholders may have a particular interest in the conservation of nature and biodiversity (e.g. Swanson, 1997; Terborgh, 1999).

Scales and stakeholders' interests. The scales at which ecosystem services are generated and supplied determine the interests of the various stakeholders in the ecosystem. Services generated at a particular ecological level can be provided to stakeholders at a range of institutional scales, and stakeholders at an institutional scale can receive ecosystem services generated at a range of ecological scales. When the value of a particular ecosystem service is assessed, different indications of its value will be found depending upon the institutional level at which the analysis is performed. For example, local stakeholders may particularly value a provisioning service that may be irrelevant at the national or international level. Hence, if a valuation study is implemented with the aim of supporting decision making on ecosystems, it is crucial to indicate on whose perspectives the values are based.

Stakeholders. A stakeholder is any entity with a declared or conceivable interest or stake in a policy concern (Schmeer, 1999). Stakeholders can be of different form, size and capacity including individuals, organizations, or unorganized groups. In most cases, stakeholders fall into one or more of the following categories: international actors (e.g. donors), national or political actors (e.g. legislators, governors), public sector agencies (e.g. MDAs), interest groups (e.g. unions, medical associations), commercial/private for-profit, nonprofit organizations (NGOs, foundations), civil society members, and users/consumers. Government institutions are stakeholders for resources in their jurisdiction, and citizens of other countries may be stakeholders when they derive welfare from the long-term indirect-benefits from ecosystem services such as carbon sequestration, tourism and nature conservation.

Stakeholders have four main attributes with respect to their interests in ecosystem services: the type of resource use practiced by the stakeholders, the level of influence (power) they hold, their degree of dependency on the ecosystem services (availability of alternatives), and the group/coalition to which they belong. These attributes can be identified through various data collection methods, including interviews with country experts knowledgeable about stakeholders or with the actual stakeholders directly. It is clear that the stakeholders deriving benefits from an ecosystem may be just as diverse as the ecosystem services themselves. Nevertheless, it is crucial to consider the differences in stakeholders when analyzing ecosystem services, as stakeholder interests and access rights will determine the interests and motivations of stakeholders in managing the resource, and management plans need to be fine-tuned with these interests in order to obtain stakeholder collaboration at different levels. This is further explained in Case study 1 below.

Case study 1. Ghana Forestry

Stakeholders are present at different institutional levels, but within each institutional level there are also major differences between stakeholders in terms of their dependency on the resource (from short term profit generator to long term livelihood dependency – compare a logging company and local forest dwellers), as well as their access rights to the resource (from traditional/customary to access rights based on negotiated contracts). This is illustrated in Figure 3 that show the stakeholders present in relation to the management of a forest system in Ghana (from Kotey et al., 1998). At the local scale, there are the forest-edge communities that harvest non-timber forest products, wood, and who benefit from the various local regulation and cultural services supplied by the forest. For instance, the forests play a role in maintaining dry season water supply from local rivers and in recharging aquifers. At the same time, local communities may exert pressure on forests in case they have an interest in converting forests to farmland, or where there is commercial harvesting of wood for charcoal production. At the district level, there are local authorities as well as people that buy the NTFP and wood from the forest in local markets. At the national level, there are national authorities, logging companies interested in short term exploitation of the forest, and a range of other groups with an interest in forest management (including NGOs, scientists, etc.). Finally, at the global level, there is an interest in the biodiversity of the forests and the carbon sequestered in it. Hence, at all levels, there are stakeholders with interests in different services supplied by an ecosystem, and the supply of these services may be compatible or not. Ecosystem services identification and stakeholder assessment provides a clear overview of the different interests in the management of the ecosystem that will have to be considered in the preparation of management strategies.

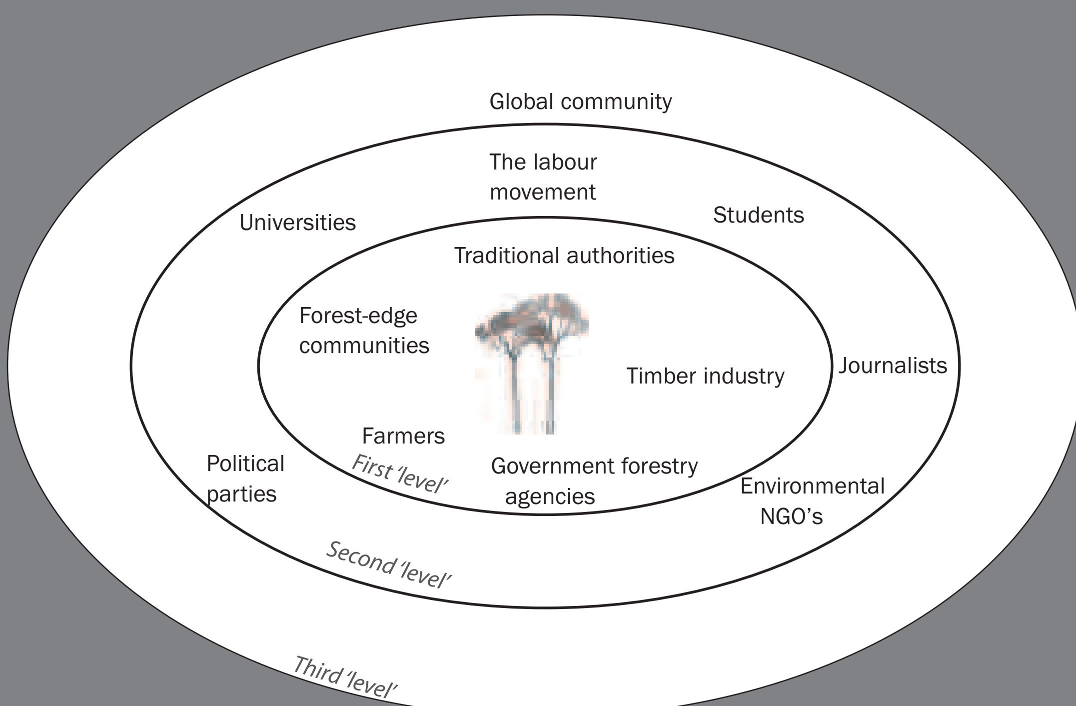

Figure 3. Levels of stakeholders in Ghana's forests (Kotey et al., 1998).

TOOL 3 Ecosystem Services Assessment

Introduction

The next step in the economic assessment is the quantification, in bio-physical units, of the relevant ecosystem services identified in the previous step. This quantification is a prerequisite for the economic valuation to be undertaken in the next step of the assessment.

For some ecosystem services, quantification is relatively straightforward. For instance, the service 'supply of firewood' involves the assessment of the amounts of firewood harvested per time unit, per area unit. Further specification may be possible, if required, by indicating the quality of the firewood (e.g. expressed as caloric value). Information on the amount of products harvested in an ecosystem may be obtained form local surveys, whereas indications on qualities of products may be obtained, for instance, from the consultation of relevant handbooks. However, for other services, quantification may be more difficult, sometimes involving limited or more extensive environmental modeling. For instance, the service 'protection from floods' that can be supplied by coastal mangroves requires analysis of the flood risks with and without the ecosystem. This can either be based on comparison of the impacts of past floods in areas with and without coastal mangroves, or it can be based on modeling of the chances of extreme, high water levels, the topography of the coastal zone, the location of population centers, and the mitigating impact of mangroves on floods.

Hence, a key step in environmental economic assessment of ecosystem services is the selection of the appropriate indicators and methodologies for the specification, in bio-physical terms, of ecosystem services. These indicators cover such aspects as the flows of products extracted from an ecosystem, the impact of the system on biochemical cycles, the impacts of regulation services on the health of people, the amount of people benefiting from the service, etc. For this step, the involvement of ecologists, hydrologists, soil scientists, etc. can be crucial in the assessment, in order to determine the exact bio-physical specifications of the ecosystem services concerned.

Purpose of the Tool

The Ecosystem Services Assessment Tool will present the user appropriate indicators for the quantification of ecosystem services. These indicators differ per ecosystem service, and several sets of indicators will be presented to the user. Furthermore, easily applicable methods will be presented that will allow the user to quantify the relevant indicators.

How to use the Tool

This tool assists the user in quantifying the selected services in biophysical terms, a prerequisite for eventual valuation of the service. The user can use Table 6 for the selection of indicators, whereas section 3.2 provides more information on a number of techniques that can be used to quantify services.

Step 3.1 Selection of indicators for ecosystem services

Before the services can be valued, they have to be assessed in bio-physical terms. For provisioning services, this involves the quantification of the flows of goods harvested in the ecosystem, in a physical unit. For most regulation services, quantification requires spatially explicit analysis of the bio-physical impact of the service on the environment in or surrounding the ecosystem. For example, valuation of the hydrological service of a forest first requires an assessment of the precise impact of the forest on the water flow downstream, including such aspects as the reduction of peak flows, and the increase in dry season water supply (Bosch and Hewitt, 1982). The reduction of peak flows and flood risks is only relevant in a specific zone around the river bed, which needs to be (spatially) defined before the service can be valued. An example of a regulation service that does usually not require spatially explicit assessment prior to valuation is the carbon sequestration service – the value of the carbon storage does not depend upon where it is sequestered. Cultural services depend upon a human interpretation of the ecosystem, or of specific characteristics of the ecosystem. The benefits people obtain from cultural services depend upon experiences during actual visits to the area, indirect experiences derived from an ecosystem (e.g. through nature movies), and more abstract cultural and moral considerations (see e.g. Aldred, 1994). Assessment of cultural services requires assessment of the numbers of people benefiting from the service, and the type of interaction they have with the ecosystem involved.

Table 6.

Indicators for the biophysical assessment of ecosystem services

Category	Key goods and services provided	Potential indicators
Provisioning services	• Food • Fodder (including grass from pastures) • Fuel (including wood and dung) • Timber, fibers and other raw materials • Biochemical and medicinal resources • Genetic resources • Ornamentals	• For all provisioning services: amount of product harvested per year; Inputs required for harvesting (time, equipment, etc.); Total inputs and outputs in case the good is used as input in a production process
Regulation services	• Carbon sequestration • Climate regulation through control of albedo, temperature and rainfall patterns	• Carbon contents of the above and below ground biomass, and in terms of soil organic matter; exchange of carbon between these three compartments and the atmosphere • Appropriate indicator for vegetation cover, e.g. Leaf Area Index or total crown cover; role of vegetation in determining moisture fluxes and temperature, resulting impacts on local and regional circulation and moisture conditions, etc.

• Hydrological service: regulation of the timing and volume of river flows	• Impact of vegetation on water flow, as a function of the topography, peak flows, vegetation cover, absorbing capacity of the soil, infiltration rates, etc. (see e.g. Bosch and Hewitt, 1982; and the case study below).
• Protection against floods by coastal or riparian systems	• Storm protective capacity depends on vegetation structure, topography, and length and width of the vegetation belt.
• Control of erosion and sedimentation	• Control of erosion and sedimentation depends on the ground cover of the vegetation, and is further a function of rainfall erosivity and soil erodibility (slope characteristics, texture, organic matter contents, etc.)
• Nursery service: regulation of species reproduction	• Nursery function depends on habitat characteristics (vegetation structure, topography) in relation to the reproduction requirements of the species involved. It can be measured in terms of numbers of juveniles produced per area unit.
• Breakdown of excess nutrients and pollution	• In particular wetland ecosystems have the capacity to filter water and recycle plant nutrients and, to some extent, absorb inorganic pollutants. The function depends on the retention time of water in the ecosystem, the temperatures affecting plant growth rates, vegetation structure, etc. It can be measured in terms of the difference in pollutant concentrations between water flowing in, and water flowing out of the system
• Pollination	• Natural vegetation may support pollination of external agricultural fields by providing a habitat for pollinators, especially bees but also other insects, bats, etc. The impact may be measured by comparing crop yields in areas with adequate pollination with crop yields in areas without adequate pollination (see e.g. the Case study below).
• Regulation of pests and pathogens	• Ecosystems may contribute to the control of certain pests and pathogens by harboring populations of species that control such pests. The impact may be measured by comparing crop yields in areas with and without such control, or health impacts in areas with an without such control.
• Protection against storms	• Ecosystems, or rows of trees, may act as windbreaks preventing wind erosion and limiting losses of crops and infrastructure from storms. This may be measured by analyzing impacts of past storms, or by modeling of erosion processes.

	• Protection against noise and dust	• Vegetation belts along highways or around industrial zones can filter air and improve air quality with regards to dust and noise. The biophysical impacts can be assessed by comparing noise levels, particulate matter levels, and concentrations of specific pollutants (e.g. NOx, S) on either side of the vegetation belt.
	• Biological nitrogen fixation (BNF)	• Through fixation of atmospheric nitrogen, leguminous plants can enhance soil fertility. Their impact can be measured in terms of soil organic matter contents.
Cultural services	• Provision of cultural, historical and religious heritage (e.g. a historical landscape or a sacred forests) • Scientific and educational information • Opportunities for recreation and tourism • Amenity service: provision of attractive housing and living conditions • Habitat service: provision of a habitat for wild plant and animal species	• For all services: amount of people benefiting from the service; type of benefits people obtain. • For the habitat service: number of species; number of red list species; hectares of ecosystem, ecosystem quality versus ecosystem in natural state, biodiversity indices.

Step 3.2 Quantitative analysis of ecosystem services

The techniques required to analyze services in biophysical terms depend entirely on the services that have been selected for the assessment. It should be noted that, in particular for regulation services, the quantification of the service is often at least as time and data consuming as the subsequent economic analysis. In addition, every service, in every economic, environmental and social context will require a specific approach with respect to the data and required approach for analysis. In the sections below, guidance is provided on approaches that can be taken for each service category.

Provisioning services. For provisioning services, surveys can reveal the flows of products harvested from the ecosystem, for instance expressed as kilograms of fruits or tons of timber harvested per time unit. It should also be examined if this flow can be extracted every year, of if this is a one time harvest in order to establish the future supply of ecosystem services. In addition, it is required to consider if the use of one service may impair the use of other ecosystem services in the future, as in the case of clearfelling of a forest.

The survey also needs to cover the efforts required to extract the products from the ecosystem. In the case of harvesting in natural forests, this relates to labor and possibly tools or equipment required for harvesting. In case the products are obtained from cultivated agricultural land, valuation should consider the inputs in the production process required to obtain the produce (as elaborated in tool 4 Valuation). This includes not only labor and equipment, but also land, fertilizers, pesticides, seeds, etc.

Regulation services. In the case of regulation services, it is important to consider the precise nature of the service supplied as well as its spatial and temporal dimensions. Table 7 provides a list of potential indicators that can be used to measure the service. The precise indicators will depend on the objective and scale of the assessment as well as the availability of data. Spatial and temporal dimensions also need to be considered. For instance, the hydrological service can be expressed as both a reduction in peak flows, and a increase in low season flow depending on the area under consideration (flood risk versus risks of seasonal water shortages). In particular the flood risk has a distinct spatial component, the flood risk will decrease with increasing distances from the water bed, depending on the topography of the valley.

Spatially explicit analysis normally requires GIS (see for examples Geoghegan et al., 1997 and Voinov et al., 1999). The spatial variation of ecological services has been elaborately studied, for instance in the fields of eco-hydrological models (e.g. Pieterse et al., 2002), and erosion and soil transport models (e.g. Schoorl et al., 2002). In general, data requirements are high for a spatially explicit approach. Initial conditions, processes, and implications of decision variables need to specified for each distinguished spatial unit. This means that assessment of spatially heterogeneous services will normally require GIS analysis, with corresponding time and budget implications.

In addition, temporal dynamics may need to be considered. For instance, the service carbon sequestration depends on the building up of carbon in either above ground biomass or as soil organic matter. Uptake depends on the growth of the vegetation, and tends to

Table 7.

Bio-physical assessment methods for regulation services

Regulation services	Assessment method
Carbon sequestration	Modeling of carbon flows in the ecosystem
Climate regulation through regulation of albedo, temperature and rainfall patterns	Regional climate models
Regulation of the timing and volume of river and ground water flows	GIS models including run off and river flow as a function of, among others, plant cover and land management
Protection against floods by coastal or riparian ecosystems	Modeling of flood risks with different vegetation cover; alternatively comparison of impacts of past floods in protected and non-protected areas.
Regulation of erosion and sedimentation	Erosion model following USLE or other models to determine erosion rates. Analysis of sedimentation rates requires catchment models of run-off and erosion, transport and deposition of sediment particles.
Regulation of species reproduction (nursery service)	Model of species reproduction, based on juveniles per successful breeding or spawning effort and the factors determining the success of reproduction (e.g. water quality, vegetation cover, etc.)
Breakdown of excess nutrients and pollution	Denitrification rates and phosphate absorbtion rates based on literature (these rates vary as a function of retention time, oxygen, iron concentrations, temperature, etc.)
Pollination (for most plants)	Pollination rates for agricultural crops can be found in literature, for non-cultivated species data is much scarcer.
Regulation of pests and pathogens	Information availability strongly dependent on the pests or pathogen involved, for some pests literature is available indicating the factors determining the chance of, and severity of outbreaks.
Protection against storms	Simple models can be used to calculate the reduction in wind speed as a function of e.g. tree cover and surface roughness. These can be translated into the wind's capacity to detach and transport particles.
Protection against noise and dust	Literature is available in order to make rough estimates of the impacts of vegetation belts on dust and air quality.
Control of run-off	USLE and other models provide indications of infiltration rates under different types of plant cover and land management (see also Bosch and Hewitt, 1982).

decrease as newly planted forests or plantations develop into mature forest stands, where there is high recycling of CO2 but much more limited net sequestration, depending on the type of forest and climatic conditions involved.

Cultural services. Cultural services are strongly dependent on the cultural backgrounds of the people that receive the service. This cultural background involves religious, moral, ethical and aesthetical motives, and they vary substantially between different societies. Ranging from indigenous to industrial societies, there are striking differences in the way cultural and amenity services are perceived, experienced, and valued by different cultures. In order to quantify the service, it is both the type of interaction and the numbers of people involved that are relevant indicators. The type of interaction ranges from frequent or occasional visits to more passive types of benefiting from the presence of a certain ecosystem, e.g. from simply knowing that the ecosystem is maintained and preserved. Prior to valuation of the service, both the type of interactions and the amount of people involved need to be analyzed.

Habitat service. In the last decades, a large number of ecological methods to quantify biodiversity and other ecological values have been developed. Wathern et al. (1986) mention that over 100 of these techniques have been described in literature. The most widely used criteria for ecological value relate to the species richness of the ecosystem, and the rarity of the species it contains, see Table 8.

Table 8.

Criteria used to measure the habitat service in 10 Ecological assessments.

Indicator	Number of studies
Diversity (including species and habitat diversity)	8
Rarity	7
Naturalness	7
Area	6
Threat of human interference	6
Representativeness	4
Uniqueness	2
Substitutability	2

Source: Margules and Usher, 1981.

Case study 2. Assessing the services supplied by forest margins

The Alternatives to Slash and Burn consortium developed a framework for comparing the impacts of different types of tropical land use on the supply of a range of ecosystems (Tomich et al., 1998). Table 9 presents the services supplied by different types of land use in tropical forest margins in the lowlands of Sumatra, Indonesia. Whereas the table does not express all services in a monetary unit, quantitative assessment of key indicators allows for comparison of the benefits supplied by different land use types. In addition, it is helpful to understand the interests of specific stakeholders. Whereas carbon storage and biodiversity are key services at the global scale, national policy makers may also be interested in the sustainability of the agricultural production capacity of the area and the returns to land, whereas local smallholder farmers will be specifically interested in the returns to labor and the household food security generated by each system. Unfortunately, the study did not consider the employment generated by each system, which will be a major indicator at both the national and local scale.

Table 9.

Comparison of services from different land use types in Sumatra, Indonesia.

Land use system	Carbon storage (aboveground tC/ha)	Aboveground plant species (# per standard plot)	Nutrient export (qualitative scale)/1	Returns from crop yields ($/ha/year)	Returns to labor ($/person-day)	Household food security (means of access)/2
Forest	306	120	0	0		
Community based forest management	120	100	0	5	4.77	$ + S
Selective logging	94	90	0	1080	0.78	$
Extensive Rubber agroforest	79	90	0	1	1.67	$
Industrial rubber forest	66	60	-0.5	878	2.25	$
Oil palm	62	25	-0.5	114	4.74	$
Upland rice / bush fallow	37	45	-0.5	62	1.47	S
Cassava / imperata	2	15	-1.0	60	1.78	$ + S

Key: /1: 0 indicates no difficulties, -0.5 minor difficulties and -1 indicates major difficulties.
/2 $ indicates that food has to be purchased on local markets and S indicates that food is supplied by the system. Note that it is not guaranteed that the cash generating systems lead to income generation for local communities, in particular in the case the forest is logged by international companies.
Source: MEA (2006)

Case study 3. Erosion control by forest systems in Western China

Hayashi et al. (2004) studied the impact of forest cover on sediment loads in the Jialingjiang River catchment (160,000 square kilometers), a tributary of the Yangtze River. They used a spatially explicit run-off and erosion model which was validated on the basis of observed daily flow rates and sediment loads of 1987. With the model, the effect of converting farmland to forest in steep slopes was examined with respect to the amount of sediment load produced in the catchment (Figure 4). Afforestation in areas in four grade classes were considered: >25%, >20%, >15%, and >10%. Farmlands with a slope value greater than 25%, 20%, 15%, and 10% cover 0.6%, 1.5%, 3.2%, and 6.3% of the Jialingjiang catchment, respectively. The assessment showed that the volume of sediment erosion decreased with afforestation, particularly for the scenarios where more area was afforested, showing afforestation to be effective for the protection of sediment production. The simulated annual total sediment loads from the whole catchment decreased up to 22% in the scenario with largest afforestation (all slopes exceeding 10%). Interestingly, the impact of afforestation in steep slopes is not much different than that of afforestation in gentler slopes. This suggests that these areas produce comparable amounts of sediment. This may be explained by the catena of the landscape. Whereas the steepest slope produce more sediment, less of the sediment ends up in the river because part of it is deposited in gentler slopes downhill. Hence, afforestation of steep slopes further away from the river, and of less steep slopes close to the river may be equally effective, and a run-off model is required to determine the precise impacts of forest cover on sediment load in different zones of the catchment.

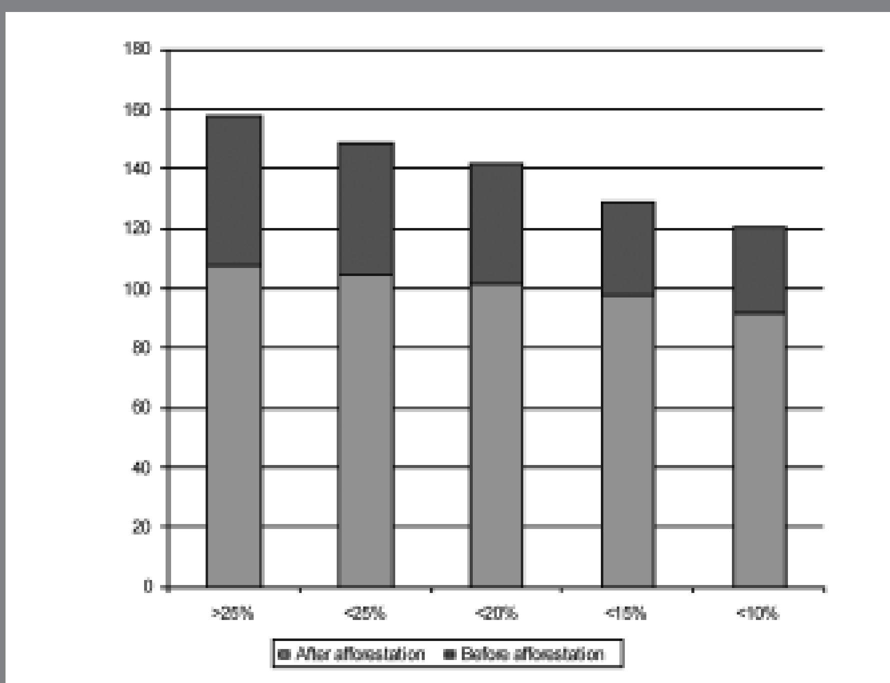

Figure 4. Impact of afforestation on different slopes on sediment loading of the Jialingjiang River (Hayashi et al., 2004).

TOOL 4 Economic Valuation

Introduction

Following welfare economics, the economic value of a resource can be determined via individual preferences as expressed by willingness to pay (WTP) or willingness to accept (WTA) for a change in the supply of that resource [1]. Aggregation of individual welfare impacts is required to obtain the welfare impact on society. Where relevant, this aggregation needs to consider equity issues, for instance where the interests of one stakeholder group (e.g. traditional ecosystem users), are considered to be more important than those of other stakeholder groups. The appropriate measure of economic value is determined by the specific context of the resources being managed. Care needs to be taken that the valuation method gives a proper indication of the value of the service involved, reflecting a true WTP or WTA, and avoiding the double counting of services or values. It is also important that the user is aware of the concepts of marginal and total value, where marginal value reflects the value of an incremental change in the supply of a resource, and total value the overall value of a resource.

There are several types of economic value, and different authors have provided different classifications for these value types. Following the Millennium Ecosystem Assessment (2003) [2], this Toolkit distinguishes the following four types: (i) direct use value; (ii) indirect use value; (iii) option value; and (iv) non-use value. They are elaborated in Table 10. The aggregated economic value of an area, combining these four value types, is often referred

Table 10.

Types of economic value.

Value Type	Description
Direct use value	This value arises from the direct utilization of ecosystems, for example through the sale or consumption of a piece of fruit. All provisioning servicess, and some cultural services (such as recreation) have direct use value.
Indirect use value	This value stems from the indirect contribution of ecosystems to human welfare. Indirect use value reflects, in particular, the type of benefits that regulation services provide to society.
Option value	Because people are unsure about their future demand for a service, they are normally willing to pay to keep the option of using a resource in the future – insofar as they are, to some extent, risk averse. Option values may be attributed to all services supplied by an ecosystem.
None-use value	Non-use value is derived from knowing that an ecosystem or species is preserved without having the intention of using it in any way. Kolstad (2000) distinguishes three types of non-use value: existence value (based on utility derived from knowing that something exists), altruistic value (based on utility derived from knowing that somebody else benefits) and bequest value (based on utility gained from future improvements in the well-being of one's descendants).

1 For details on the concept of WTP and WTBC, and when they converge to the same value, the reader is referred to basic environmental economics textbooks such Freeman (1993) or Perman et al. (1999).

2 For different classifications of economic value types, the reader is referred to, for example, Hanley and Spash (1993) and Kolstad (2000)

Table 11.

Value types in relation to ecosystem services.

Direct use value	Indirect use value	Option value	Non-use values
Food	Climate regulation and carbon sequestration	Potential future uses	Habitat service and biodiversity
Fodder and grass from pastures	Hydrological service	Future value of information	Cultural heritage
Fuel and other energy	Protection against floods		
Timber, fibers and raw materials	Control of erosion and sedimentation		
Biochemical and medicinal resources	Nursery service		
Genetic resources	Breakdown of excess nutrients and pollution		
Ornamentals	Pollination		
Recreation	Regulation of pests and pathogens		
Scientific & education information	Protection against storms		
	Protection from noise & dust		
	Biological nitrogen fixation		

Source: adapted from Pearce and Turner (1990) and Barbier et al. (1997)

to as Total Economic Value (TEV). Table 11 indicates the value type most commonly associated with specific ecosystem services.

Note that these different values may or may not be reflected in a market value. In most cases, a significant part of the Direct Use Value will be reflected in market transactions, but most of the other value types will not. They may not be reflected in market transactions because, for instance, they have a public goods character, or because a market has not (yet) been established for the service. Because of the economic benefits they provide, the non-market economic values also need to be included in economic Cost-benefit assessment. Furthermore, several authors have discussed or analyzed the non-economic value of ecosystems. These non-economic values are independent of any human use or interaction with an ecosystem ("what is the value of a tropical forest if there were no people on the planet ?"). This is expressed in figure 5. Although such ecocentric values may exist, this Toolkit includes only anthropocentric economic values, including both market and non-market values.

In case future and present benefits have to be compared, discounting is required. In some cases, in particular where a long time horizon is selected for the assessment, the discount rate can have a major impact on the economic cost benefit analysis of land management options. This is particularly relevant where sustainable approaches are promoted which may lead to a higher supply of ecosystem services in the future compared to currently used land management practices. In these cases, the selection of the discount rate is a crucial factor in the analysis, as discussed in Section 4.3

Figure 5. Market versus non-market values. This Toolkit focuses on market and non-market economic values only.

Purpose of the Tool

The purpose of this tool is to guide the user in analyzing the total economic value of the services supplied by an ecosystem. The Tool presents the different valuation techniques, and for which services, and in which contexts, the valuation techniques are appropriate. The Tool also informs the user of the various limitations of economic valuation approach, and provides guidance on the interpretation of the outcomes of valuation studies.

Contents of the Tool

The Tool contains three steps, dealing with: (i) selection of the value indicator; (ii) the selection of the actual economic valuation technique; and (iii) choosing the discount rate. In the second Step (4.2), a description of the most important economic valuation methods is provided, and references are provided where the user can find more detailed information on each of the valuation methods. Note that Appendix 2 provides an overview of useful internet addresses, many of which also provide information on economic valuation techniques.

Step 4.1 Selection of value indicators

According to welfare economics, the welfare generated by an ecosystem service, or the economic value of this service, is the (weighted) sum of the utility gained by all individuals as a result of the provision of the ecosystem service. Utility is gained by the person consuming the ecosystem service (e.g. by eating a piece of fruit or walking in a national park). However, utility cannot be measured directly. In order to provide a common metric in which to express the benefits of the widely diverse variety of services provided by ecosystems, the utilitarian approach usually attempts to measure all services in monetary terms (MEA, 2003).

Changes in welfare are reflected in people's willingness to pay (WTP) or willingness to accept (WTA) compensation for changes in their level of use of a particular good or bundle of goods (Hanemann 1991). Although WTP and WTA are often treated as interchangeable [3], there are important conceptual and empirical differences between them. In general, 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 higher than WTP. In the case of private goods traded in perfectly functioning markets, the willingness-to-pay for a good is reflected in the price paid for that good on the market. However, clearly, most ecosystem services are not traded in a market, and other ways need to be sought in order to reveal people's willingness-to-pay for the service (as elaborated in Step 4.2).

In order to determine the societal value of the service, it is required to analyze the economic surplus generated by the service. Whereas WTP for an increase in ecosystem service supply may be low at current supply levels, this WTP may strongly increase in case of a shortage of the service (compare the price of water during a period with ample supply with it's price during a drought). Two central concepts here are the consumer and the producer surplus, as described below.

(i) The consumer surplus. The concept of consumer surplus was first described by Dupuit and introduced to the English speaking world by Marshall (in 1920): 'The excess of price which a consumer would be willing to pay rather than go without the thing, over that what he actually pays is the economic measure of this surplus of satisfaction' (Johansson, 1999). Hence, the market price plus the consumer surplus equals the utility of a specific good for a certain consumer. Note that the utility gained by the consumer from an actual transaction also depends upon a number of other factors, such as transaction costs. Estimation of the consumer surplus generally requires the construction of a demand curve. For more details, for instance on sources of inconsistency in ordinary demand curves and solutions for these inconsistencies, see e.g. Willig (1976), Freeman (1993) or Perman et al. (1999).

(ii) The producer surplus. The producer surplus indicates the amount of welfare a producer gains at a certain production level and at certain price. The estimation of the producer surplus generally requires the construction of a supply curve (see e.g. Perman et al., 1999). In the short term, a producer's fixed costs can be considered foregone. Hence, in micro-economics, the individual producer surplus is defined as total revenues minus variable costs (Varian, 1993). In the valuation of ecosystem services, the producer's surplus needs to be considered if there are costs related to "producing" the ecosystem good or service (Freeman, 1993; Hueting et al., 1998). In general, in the case of private ecosystem good or services, these costs relate to the costs of harvesting or producing the ecosystem good or service. The supply curve will in many cases show a relatively steep increase at higher quantities of ecosystem service supplied – e.g. the costs of providing marginal cleaner water increase as purity becomes higher (Hueting, 1980).

Whereas the WTP reflects the marginal value of an ecosystem service, consumer and producer surpluses represent the total societal benefits generated by an ecosystem service. However, as data is not always available to calculate these surpluses, other

3 For details on the concept of WTP and WTBC, and when they converge to the same value, the reader is referred to basic environmental economics textbooks such Freeman (1993) or Perman et al. (1999).

indicators of economic value have also been used in valuation studies.

Some studies have also suggested that market value (price times quantity) of certain products provide an indication of the value. However, this approach is not recommendable, as it does not account for the investments that need to be made in order to obtain the produce involve (e.g. the costs of boats, nets and labor of the fishermen).

For those services that can be translated into a monetary value, a choice needs to be made if the values are expressed as benefits per year (recommendable in case few changes in services supply, and only limited price increases or decreases can be expected), or if the streams of benefits are expressed as a Net Present Value, which indicates the current present value of the net present and discounted future flows of services (see Box 2). In the last case, the user has to select a discount rate, as specified in Step 4.3 ('Selection of the Discount rate').

Step 4.2 Valuation

Following neo-classical welfare economics, valuation requires analysis and aggregation of the consumer and producer surpluses (Freeman, 1993). In the last 3 decades, a range of economic valuation methods for ecosystem services has been developed. They differ for private and public goods.

(i) Valuation of private goods. In the case of private goods or services traded in the market, price is the measure of marginal willingness to pay and it can be used to derive an estimate of the economic value of an ecosystem service (Hufschmidt et al., 1983; Freeman, 1993). The appropriate demand and supply curves for the service can - in principle - always be constructed. However, in practice this is often difficult, as (i) it is not always known how people will respond to large increases or decreases in the price of the good, and (ii) it may be difficult to assess when consumers will start looking for substitute goods or services. In case of substantial price distortions, for example because of subsidies, taxes, etc., an economic (shadow) price of the good or service in question needs to be constructed. In some cases, this can be done on the basis of the world market prices (Little and Mirrlees, 1974; Little and Scott, 1976). In case the private good is not traded in the market, because it is bartered or used for auto-consumption, shadow prices need to be constructed on the basis of: (i) the costs

BOX 2. Calculating the Net Present Value

The Net Present Value (NPV) depends on the flows of net benefits in year t, now and in the future (Ct), the discount period considered (T) and the discount rate (r) according to the formula below. Note that discounting leads to a rapid decline in the importance of future benefits, e.g. a dollar obtained 100 year from now is, at a 2% discount rate, worth only 0.14 cents now. This means that discounting is not easily compatible with the notion of sustainable management, where the interests of future generation are believed to be on par with our current interests. Therefore, in particular for longer term issues such as climate change, simple discounting procedures are no longer favoured, and the use of zero discount rates, or discount rates that decrease over time towards zero, have been proposed (see e.g. Pearce et al., 1990 for more information).

$$NPV = \sum_{t=0}^{T} \frac{Ct}{(1+r)^t} = C_0 + \frac{C_1}{(1+r)^1} + \frac{C_2}{(1+r)^2} + \frac{C_3}{(1+r)^3} + \dots$$

of substitutes; or (ii) the derived benefit of the good (Munasinghe and Schwab, 1993).

(ii) Valuation of public goods. Two types of approaches have been developed to obtain information about the value of public ecosystem services: the expressed and revealed preference methods (Pearce and Howarth, 2000). These methods have also been called direct and indirect valuation methods, respectively.

With *expressed valuation methods,* either market prices or various types of questionnaires are used to reveal the willingness-to-pay of consumers for a certain ecosystem service. The most important direct approaches are the Contingent Valuation Method (CVM) and related methods. In the last decades, CVM studies have been widely applied (see e.g. Nunes and van den Bergh, 2001 for an overview). It is the only valuation method that can be used to quantify the non-use values of an ecosystem in monetary terms. Information collected with well-designed CVMs has been found suitable for use in legal cases in the U.S. - as in the case of the determination of the amount of compensation to be paid after the Exxon Valdez oil spills (Arrow et al., 1993).

The *revealed preference methods* use a link with a marketed good or service to indicate the willingness-to-pay for the service. There are two main types of revealed preference methods:

- **Physical linkages.** Estimates of the values of ecosystem services are obtained by determining a physical relationship between the service and something that can be measured in the market place. For instance, with the damage-function (or dose-response) approach, the damages resulting from the reduced availability of an ecosystem service are used as an indication of the value of the service (Johanson, 1999). This method can be applied to value, for instance, the hydrological service of an ecosystem.

- **Behavioral linkages.** In this case, the value of an ecosystem service is derived from linking the service to human behavior – in particular the making of expenditures to offset the lack of a service, or to obtain a service. An example of a behavioral method is the Averting Behavior Method (ABM). There are various kinds of averting behavior: (i) defensive expenditure (a water filter); (ii) the purchase of environmental surrogates (bottled water); and (iii) relocation (OECD, 1995; Pearce and Howarth, 2000). The travel cost method and the hedonic pricing method are other revealed preference approaches using behavioral linkages.

An overview of the main valuation methods, and the value types they can be used for, is presented in Table 12. The remainder of this section provides a detailed description of six main ecosystem valuation techniques: (i) replacement cost method; (ii) averting behavior method; (iii) travel cost method; (iv) production factor approach; (v) hedonic pricing; and (vi) contingent valuation (CVM). The first five methods are revealed preference valuation methods, the last (CVM) is a stated preference valuation method.

In case there are no resources or data to allow for an actual valuation of the ecosystem services supplied by an area, it is possible to use a so-called benefit transfer approach, i.e. using value indications from other areas as

Table 12.

Valuation methods and their applicability to different value types (Based upon Pearce and Turner, 1990; Hanley and Spash, 1993; Munasinghe, 1993; Cummings and Harrison, 1995).

Valuation method	Suitable for	Value category			
		direct use value	indirect use value	option value	non-use value
Indirect methods:					
1) averting behavior method	Applicable to services that relate to the purification services of some ecosystems.	x	x		
2) travel cost method	Can be used to value the recreation service.	x			
3) production factor approach	Applicable where ecosystem services are an input into a production process	x	x		
4) hedonic pricing	Applicable where environmental amenities are reflected in the prices of specific goods, in particular property.	x	x		
Direct methods:					
5) CVM	The use of CVM is limited to goods and services that are easily to comprehend for respondents – excluding most regulation services	x		x	x
6) market valuation	Ecosystem goods and services traded on the market	x	x	x	

described in literature and transfer these to the ecosystem under consideration. Whereas this method is is a fast method of obtaining economic information on ecosystem services, it has a number of important drawbacks, in particular its high degree of uncertainty. Box 3 provides further information on the application of benefit transfer techniques.

4.2.1 The Averting Behavior Methods

The Averting Behavior Methods (ABM) consider expenditures made to avert or mitigate negative effects from environmental degradation. ABM relies on the assumption that people perceive the negative effects of environmental deterioration on their welfare and that they are able to adapt their behavior to avert or reduce these effects. This means, for instance, that people understand the negative effects of ozone depletion and that they will buy products such as hats and suncream to prevent damage to their health. The willingness to pay for a clean environment is calculated on the basis of people's purchases of products and services to avert the negative effects of pollution.

The Averting Behavior Methods comprises three different sets of methods: (i) damage costs avoided; (ii) preventive expenditure; and (iii) replacement costs methods. For example, people can respond to a reduction in tap water quality by (i) become sick and incur health costs; (ii) installing a water filter at home; or (iii) buy bottled water. Where the damage costs or the preventive or replacement expenditures are reflected in market transactions, this method provides an indirect way of analyzing environmental benefits including the supply of ecosystem services. Market data can often relatively easily be collected, and in this case data on averting behavior of affected stakeholders can and used to obtain an estimate of the value of the service.

ABM is a cost-based method, using the costs of purchased items to value environmental qualities. However, the social preferences for a healthy environment may be much greater than the expenditures on these products and, since the market prices of products are used to value the environment, this method does not capture the consumers' surplus. Instead, it is assumed that the costs of avoiding damages or replacing natural assets or their services provide useful estimates of the value of these assets or services. This is based on the assumption that, if people incur costs to avoid damages caused by lost ecosystem services, or to replace the services of ecosystems, then those services must be worth at least what people paid to replace them. This assumption may or may not be true. However, in some cases it may be reasonable to make such assumptions, and measures of damage cost avoided or replacement cost are generally much easier to estimate than people's willingness to pay for certain ecosystem services. In general, the methods are most appropriately applied in cases where damage avoidance or replacement expenditures have actually been, or will actually be, made.

Potential caveats

ABM presupposes that people are fully aware of the potential impacts of changes in environmental quality, and that they have the option of responding to this by their individual behavior or through individual decisions to purchase preventive measures. However, often not all people affected will fully inform themselves on the potential impacts of a reduced supply of ecosystem services or environmental quality. Furthermore, people may not react to small changes in environmental quality, but respond only when certain thresholds have been passed.

Further reading

Pearce and Turner (1990) and Freeman (1993) describe the theoretical aspects of the ABM. Champ and Brown (2003) provide an overview of applications, and Young (2005) describes a number of applications of ABM to value water resources.

4.2.2 Travel Cost Method

The travel cost method is used to estimate the economic use value of the recreation service of ecosystems. The basic premise of the travel cost method is that the time and travel cost expenses that people incur to visit a site is an indicator for the willingness to pay of people to visit the site. The method can be used to estimate the economic benefits or costs resulting from: (i) changes in access costs for a recreational site ; (ii) elimination of an existing recreational site; (iii) changes in environmental quality at a recreational site; or (iv) addition of a new recreational site. There are two basic approaches in applying the TCM. The first is the simple zonal travel cost approach, the second the individual travel cost approach which uses a more detailed survey of visitors.

Application of the Zonal Travel Cost Approach. The zonal travel cost method is the simplest and least expensive approach. It is used to estimate a value for recreational services of the site as a whole. The zonal travel cost method is applied by collecting information on the number of visits to the site from different distances. In order to determine the willingness to pay of visitors, distance circles are drawn around the site. The TCM assumes that people in all circles have homogeneous preferences. This information is used to construct the demand function for the site, and estimate the consumer surplus, or the economic benefits, for the recreational services of the site. The method consists of 7 basic steps, see Table 13.

Table 13.

The zonal TCM

Step	Analysis
1	Define a set of zones surrounding the site. These may be defined by concentric circles around the site, or by geographic divisions such as metropolitan areas or counties surrounding the site at different distances.
2	Collect information on the number of visitors from each zone, and the number of visits made in the last year.
3	Calculate the visitation rates per 1000 population in each zone. This is simply the total visits per year from the zone, divided by the zone's population in thousands.
4	Calculate the average round-trip travel distance and travel time to the site for each zone. Assume that people in Zone 0 have zero travel distance and time. Each other zone will have an increasing travel time and distance. Next, using average cost per kilometer and per hour of travel time, calculate the average travel cost per trip per zone.
5	Estimate, using regression analysis, the equation that relates visits per capita to travel costs. From this, the researcher can estimate the demand function for the average visitor.
6	Construct the demand function for visits to the site, using the results of the regression analysis. The first point on the demand curve is the total visitors to the site at current access costs (assuming there is no entry fee for the site). The other points are found by estimating the number of visitors with different hypothetical entrance fees (assuming that an entrance fee is viewed in the same way as travel costs).
7	Estimate the total economic benefit of the site to visitors by calculating the consumer surplus, or the area under the demand curve.

Application of the Individual Travel Cost Approach. The individual travel cost approach is similar to the zonal approach, but uses survey data from individual visitors in the statistical analysis, rather than data from each zone. This method requires more data collection and a slightly more complicated analysis, but will give more precise results because it allows to correct for heterogeneity among visitors within the distance circles. Survey questions can include: (i) location of the visitor's home – how far they traveled to the site; (ii) how many times they visited the site in the past year or season; (iii) the length of the trip; (iv) the amount of time spent at the site; (v) travel expenses; (vi) the person's income or other information on the value of their time; (vii) other socioeconomic characteristics of the visitor; (viii) other locations visited during the same trip, and amount of time spent at each; (ix) other reasons for the trip (is the only purpose of the trip to visit the site, or are there additional purposes)

Using the survey data, the researcher can proceed in a similar way to the zonal model, by estimating, using regression analysis, the relationship between number of visits and travel costs and other relevant variables. However, this time, the researcher uses individual data rather than data for each zone to estimate the demand function. The regression equation yields the demand function for the "average" visitor to the site, and the area below this demand curve gives the average consumer surplus. This is multiplied by the total relevant population (the population in the region where visitors come from) to estimate the total consumer surplus for the site.

Potential caveats in application of the TCM

The main advantage of the TCM is that it provides a theoretically correct approach to value recreational services accruing to visitors of a site. Some caveats are that it is sometimes difficult to correct for multiple purpose travels, when not all travel costs are made to visit the site under consideration. In addition, visitors to a park may stay in a nearby holiday house, whereas part of the reason for going to the holiday house is the presence of the park. There may also be a skewed distribution in the demand function where parks in developing countries attract both national and international visitors (in this case it is recommendable to construct two different demand functions).

Further reading

For more information on the theory of the TCM, the reader is referred to, for instance, Hanley and Spash (1993), Haab and McConnell (2000) and Mäler and Vincent (2005). Aylward and Lindberg (1999) provide an application of the TCM in Costa Rica to calculate the price sensitiveness of visits to national parks in the country. Seenprachawong (2003) compares application of TCM and Contingent Valuation to value coral reefs in Thailand. In addition, the website *www.ecosystemvaluation.org* provides a detailed description plus a case study of application of the TCM.

4.2.3 Production factor methods

Production factor methods have also been called 'factor income method' or 'productivity method'. They are used to estimate the economic value of ecosystem products or services that contribute to the production of commercially traded goods. The method can be applied when ecosystem services, along with other inputs, are used to produce a marketed good. For example, the benefits of supplying water for irrigation purposes can be derived from the increased production as a consequence of the irrigation.

If a natural resource is a factor of production, then changes in the quantity or quality of the resource will affect the production costs of the marketed goods involved. This has two welfare effects. First, if the quality or price to consumers of the final good changes, there will be changes in consumer surplus. Second, if productivity or production cost changes, producers will be affected and there will be changes in the producer surplus. Hence, in principle, where ecosystem services are used as input in a production process, the economic benefits from changes in ecosystem services supply can be estimated using changes in observable market data.

Application of Production Factor Methods consists of a two-step procedure. First, the physical effects of changes in a biological resource or the supply of an ecosystem service on an economic activity need to be assessed. Second, the impact of these environmental changes should be valued in terms of the corresponding change in the marketed output of the corresponding activity. This means data must be collected on how changes in the quantity or quality of the natural resource affect (i) the costs of production for the final good; (ii) the supply and demand for the final good; and (iii) the supply and demand for other factors of production (e.g. reduced irrigation water availability may affect the demand for fertilizers). This will normally involve the construction of supply and demand curves for the good involved. In this way, the analysis reveals how changes in ecosystem services supply affect the consumer surplus (as a function of lower or higher prices for the marketed good) and/or the producer surplus (as a function of the increased costs to producers and their capacity to grow alternative crops).

Note that it is not always necessary to construct supply and demand curves, in which case the application of the method become much more straightforward and much less data intensive. For instance, if the method is applied to value the supply of an ecosystem service that affects only a small part of the production of a certain good traded on a market, it can be assumed that the ecosystem service does not affect the consumer surplus. The specific condition here is that a change in the supply of the ecosystem service does not change the price (or quantities) at which the good concerned is available on the market. For instance, if the pollination service is valued, valuation of pollination at the scale of the individual farm does not need to consider any price effects and consequent changes in consumer surplus. If, on the other side, pollination is valued at the scale of the country (e.g. to estimate the importance of pollination for national agriculture), such price effects can be expected and need to be calculated. In the 'simple' case of the individual farmer, pollination effects the producer surplus of the individual farmer only, and the economic value can be calculated by multiplying impacts on production times the net farm-gate economic price.

A second example of when the method is easily applied is where the ecosystem service in question is a perfect substitute for another input in the production process. For example, increased water quality in a reservoir may mean that less chlorine is needed for treating the water. Thus, in this example, the benefits of increased water quality can be directly measured by the decreased chlorination costs. Note that if this would lead to large changes in the market price of the final product, there would be an additional benefit to consumers and, hence, a change in consumer surplus.

Hence, the method is highly useful to estimate the wide range of ecosystem services that provide an input in the production process including various provisioning services such as water, timber, bamboo, etc. as well as regulation services including pollination and biological nitrogen fixation. However, unless changes in the supply of the ecosystem service do not lead to changes in market prices in the final product, the method is data intensive as supply and demand curves need to be constructed.

Potential caveats

A potential bias of the PFM is that effects on production may have been distorted by averting behavior. For instance, producers will try to avert the effects of reduced natural qualities by undertaking prevention activities, such as shifting to different crops or products, adapting cultivation or harvesting techniques. Because PFM is based on dose-response relations which involve a considerable amount of ecological information and because it requires economic data on natural products as well, it has a large data requirement, in particular if it is necessary to account for demand and supply dynamics in the valuation of responses.

Further reading

Freeman (1993) gives an account of the theoretical foundation underlying the Production Factor Method. Ricketts et al. (2004) provide a simple method to calculate the value of the pollination service at the local scale (as also elaborated in Case Study 4). Southwick and Southwick (1992) calculate the consumer surplus related to the pollination service with respect to crop pollination in the USA, and Gordon and Davis (2003) examined the producer surplus of honey bee pollination in relation to Australian agriculture. Cooke (1998) examines local production factors including ecosystem services on agricultural production in Nepal.

4.2.4 The Hedonic Pricing Method (HPM)

The hedonic pricing method is used to estimate economic values for those ecosystems or ecosystem services that directly affect market prices. Hedonic pricing methods can be used to estimate economic benefits or costs associated with (i) environmental quality, including air, water and noise pollution; and (ii) environmental amenities, such as aesthetic views or proximity to recreational sites. However, it is most often used to value environmental amenities that affect the price of residential properties. For example, the price of a house in quiet and beautiful surroundings is likely to be higher than the price of the same kind of house next to a highway. HPM starts with a regression of house prices against all their relevant characteristics. This leads to a hedonic price function of the following shape: Value(house) = f(size, style, garden size, age, environmental characteristics, etc.). From this function one can calculate the willingness to pay for a marginal change in each of these explaining

variables. This is the implicit price of the amenity under investigation. From these implicit prices, the demand curve for a specific amenity can be derived. The demand curve is then used for estimating the economic value of an amenity such as natural beauty.
HPM has a large data requirement because both primary data (characteristics of the surroundings) and secondary data (market transactions) need to be collected. For instance, the value of a house or wage depends on many factors: there are social factors, such as employment opportunities, taxes and accessibility, and data need to be gathered for all these factors.

Potential caveats

Since the number of explaining variables can be numerous, there is a risk that not all important variables are included in the regression analysis. It is also possible that there are several amenities that influence the price of a house in opposite directions. There may, for example, be a positive influence of a park nearby, but at the same time two noxious facilities which supply jobs. Finally, the house market may be distorted due to governmental interventions which leads to a bias in the assessment of the economic value of amenity ecosystem services as well (Pearce and Markandya, 1989).

Further reading

Freeman (1993) examines the theories behind hedonic pricing. Le Goffe (2000) examines the application of HPM to examine externalities of agriculture and forestry in the USA, and Kim et al. (1998) apply HPM to measure the benefits of air quality improvement in the USA. Two examples from developing countries are Shanmugam (2000) who estimates values of life and health, using data from India, and Macedo (1998) who examines the Belo Horizonte housing market in Brazil with HPM.

4.2.5 The Contingent Valuation Method (CVM)

CVM is a survey method in which respondents are asked how much they are willing to pay for the use or conservation of an ecosystem service. Their stated preferences are assumed to be contingent upon the alternative goods that are offered in a 'hypothetical market'. The three main elements of a CVM are: (i) a description of the ecosystem service to be valued; (ii) a description of the payment vehicle; and (iii) a description of the hypothetical market (Ruijgrok, 2004). The payment vehicle explains how and to whom the money will be paid. One can pay for a good in cash every time it is used or by means of an increased income tax. The description of the hypothetical market should include an identification of who will provide and who will pay for the ecosystem service. It should be made clear that the payment is a collective action; everybody else will also pay, otherwise respondents may refuse to pay although they appreciate the good. In order to prevent overestimates, respondents should also be reminded of the possibility of spending their income on goods other than nature. CVM measures benefit-based preferences and it includes the consumers' surplus.

CVM, and related Choice experiments, are the only methods available to value non-use values of ecosystems, and they can also be used for selected other services and value types. In general, CVM is an appropriate economic valuation method for environmental goods that have no indirect effects on other goods. It is therefore suited for the valuation of amenities or other easy to perceive aspects of nature, but not for the valuation of natural processes, such as climate regulation, where effects on human welfare are difficult to grasp for respondents to a questionnaire. In general, CVM does not produce valid measurements when it concerns goods that people are not familiar with. Nor does it work when people reject responsibility for the good in question. If people are asked, for example, about their willingness to pay for clean soil, they may state that it is zero, because they feel the polluter should pay. This does not mean that they do not appreciate clean soil. One may also remark here that it is better to value goods that have an international character on a cost basis, because in a CVM-survey respondents will not know what to answer if they realize that reducing pollution in their own country does not solve the global pollution problem, if the other countries do not make an effort too.

Choice experiments

Choice experiments are strongly related to contingent valuation, in that it can be used to estimate economic values for a broad range of ecosystem services, for both non-use and well as use values. Like contingent valuation, it is a hypothetical method – it asks people to make choices based on a hypothetical scenario. However, it differs from contingent valuation because it does not directly ask people to state their values in dollars. Instead, values are inferred from the hypothetical choices or tradeoffs that people make. The contingent choice method asks the respondent to state a preference between one group of environmental services or characteristics, at a given price or cost to the individual, and another group of environmental characteristics at a different price or cost. Because it focuses on tradeoffs among scenarios with different characteristics, contingent choice is especially suited to policy decisions where a set of possible actions might result in different impacts on natural resources or environmental services. For example, improved water quality in a lake will improve the quality of several services provided by the lake, such as drinking water supply, fishing, swimming, and biodiversity. In addition, while contingent choice can be used to estimate dollar values, the results may also be used to simply rank options, thout focusing on dollar values.

Potential caveats

There are two main points of criticism against CVM. First, CV estimates are sensitive to the order in which goods are valued; the sum of the values obtained for the individual components of an ecosystem is often much higher than the stated willingness-to-pay for the ecosystem as a whole. Second, CV often appears to overestimates economic values because respondents do not actually have to pay the amount they express to be willing to pay for a service (see e.g. Diamond and Hausman, 1994 and Hanemann, 1995).

Further Reading

For more information on the theories of the TCM, the reader is referred to, for instance, Haab and McConnell (2000). Diamond and Hausman (1994) present an overview of the various critiques on the CVM. Arrow et al. (1993) present the well-known application of CVM to support the damage claims following the Exxon Valdez oil spills. Whittington (1998) reviews the application of CVM in developing countries. Finally, Boxall et al. (1996) compares the different approaches to CVM.

BOX 3. Benefit transfer

If few data are available for an ecosystem, crude estimates of the values of ecosystem services may be obtained through 'benefit transfer' - the transfer of ecosystem values to settings other than those originally studied (Green et al., 1994; Willis and Garrod, 1995; and Brouwer et al., 1997). Costanza et al. (1997) and Pearce and Pearce (2001) provide indications of the values of a range of ecosystem services in selected ecosystems (see the table below). However, there are severe limitations with respect to the application of this methodology. The potential value of a service varies widely as a function of the type of ecosystem involved and it's socio-economic and bio-physical setting. This is reflected in the large spread in the values indicated in the table. The user is recommended to be cautious with the use of benefit transfer, and use it only if there are reasonable assumptions to assume that the ecosystem type, and it's socio-economic and bio-physical setting are comparable.

Value ranges for ecosystem services

Ecosystem Service	Value range (US$/ha/year)
Provisioning services	
Food	6–2761
Raw materials	6–1014
Regulation services	
Carbon sequestration	7–265
Climate regulation through control of albedo, temperature and rainfall patterns	88–223
Hydrological service: regulation of the timing and volume of river discharges	2–7500
Control of soil erosion and sedimentation	10-250
Nursery service	142–195
Breakdown of excess nutrients and pollution	50 – 20,000
Pollination	14-25
Regulation of pests and pathogens	2-78
Cultural services	
Tourism and recreation	2-3000
Cultural and historical heritage	1-1500
Ammenity services including pleasant living conditions	75-10,300
Habitat service	3–1523

Step 4.3 Selecting the discount rate

If the value of ecosystem services is expressed as NPV (instead of as an annual flow), the discount rate is a crucial factor. Discounting is used to compare present and future flows of costs and benefits derived from the ecosystem. Often, unsustainable exploitation of ecosystems involves high short term benefits (e.g. clear-cut of the timber stands) whereas sustainable management leads to a more long term flow of benefits (e.g. through a sustainable harvesting regime). Selection of a high discount rate implies that future costs and benefits are not deemed very important and, in this case, an economic analysis may indicate that it is efficient to immediately harvest all stands of timber in a forest, even if this would lead to an irreversible loss of ecosystem services for future generations.

The discount rate can be derived following two approaches, on the basis of (i) the consumption discount rate; and (ii) the social opportunity costs of capital (Pearce and Turner, 1990). The consumption discount rate indicates, among others, that most people prefer immediate rather than future consumption, and the social opportunity costs of capital represent the rate of return on capital. In a simple economy with no taxes or inflation, and perfect capital markets, the consumption discount rate equals the social opportunity costs of capital equals the market interest rate (Lind, 1982; Varian, 1993). In reality, this is usually not the case. For instance, due to taxation and inflation, the market interest rate is higher than the consumption discount rate (Freeman, 1993; Hanley and Spash, 1993). Hence, a choice needs to be made regarding the discount rate to be used (Pearce and Turner, 1990).

The discount rate to be used in environmental cost-benefit analysis is still subject to debate (e.g. Howarth and Norgaard, 1993; Norgaard, 1996; Hanley, 1999). For instance, Freeman (1993) indicates that the discount rate, based upon the after- tax, real interest rate, should be in the order of 2 to 3% provided that the streams of benefits and costs accrue to the same generation, whereas Nordhaus (1994) argued that a 6% discount rate is most consistent with historical savings data. In practice, in many valuation studies, a 5% discount rate has been used.

Note that discount rates in the order of 2 to 5% still lead to rapid depreciation of future costs and benefits. At a discount rate of 2%, the value of US$ 1 in 100 years amounts to not more than 14 cents. Hence, through discounting, a much larger weight is attached to the net benefits accruing to current generations as compared to the benefits for future generations. Therefore, it is highly disputed if discounting is appropriate for the analysis of environmental issues with large time lags between investments in mitigation measures and positive economic impacts such as climate change. In general, the use of a high discount rate will favor ecosystem management options that lead to relatively fast depletion of resources, whereas a low discount rate will stress the economic benefits of more sustainable management options (Pearce and Turner, 1990; Tietenberg, 2000).

Case study 4. Value of the pollination service in coffee plantations

The pollination service is an important regulation service supporting global agriculture. A number of recent studies have attempted to measure the economic value of the service in different agro-ecological and economic settings. This case studies deals with the local value of pollination in coffee cultivation. The case refers to C. arabica, which accounts for over 75% of the global coffee production. The contribution of pollination to coffee production has been shown in a range of studies. For instance, through statistical analysis of coffee yields before and after the introduction of African honey bee in the neotropics in the early 1980s, Roubik (2002) analyzed the impact of pollination on coffee production. Roubik (2002) estimates that pollination of coffee plants (all insects) increases global C. arabica yields by on average some 36%. Furthermore, Klein et al. (2003) show that a loss of the pollination service led to a 12.3% lower yield in Indonesian C. arabica plantations. Ricketts et al. (2004) found that enhanced pollination of Costa Rican coffee plants near forest edges led to a 20.8% higher yield in comparison with coffee plants in the centre of the field.

Ricketts et al. (2004) provide a simple method to calculate the value of the pollination service for a large coffee producer in the Valle General, Costa Rica. The plantation comprises both sites located close to remaining patches of natural forest, and sites further away form natural forest. The forest patches provide a habitat to non-native honey bees as well as 10 native species of Meliponini stingless bees. Ricketts et al. (2004) show that the bees have difficulties reaching the parts of the coffee plantation located farthest from the forest, and establish that bee pollination makes an important contribution to coffee yields. The formula that they use to calculate the economic benefits of the pollination service at the local scale is:

$$W = S \cdot \Delta q \cdot (p - c)$$

with W = benefits for the farmer;
S = area
¢q = increase in production as a consequence of pollination
p = farm-gate coffee price
c = variable costs related to coffee harvest.

In the Costa Rican study, 480 ha of coffee fields (S) are close (<1 km) to two patches of forest that have been conserved on the plantation, the increase (¢q) in coffee is 20.8% x 14.240 kg/ha, the farm-gate price (p) is US$ 0.071 /kg, the labor costs of harvesting (c) are US$ 0.028 /kg, and the resulting value (W) of the two patches of forest that maintain pollinator populations that cater the coffee plantation is US$ 62,000. This represents 7% of the annual income of the plantation (Ricketts et al., 2004). This example demonstrates that pollination can make an important economic contribution at the scale of the individual plantation. Note that it does not provide sufficient guidance for the management of the pollination service, it can not be derived from this experiment how much forest patches need to be preserved in order to maintain the pollination service in the plantation; either more (if not all coffee fields are sufficiently pollinated) or less (if populations could do with smaller habitats) forest patches could be optimal for the farmer.

Case study 5. Valuation of the tourism service

The tourism service of ecosystems generates benefits for both visitors to areas of natural beauty and the tourism sector including airline companies, hotels, local service providers and shops. The value of the service for visitors is represented by the consumer surplus they obtain from their trip. For the individual visitor, this is derived from the willingness to pay for the trip minus the costs of the trip. Through a Travel Cost method, the surplus for the visitor can be calculated.
The benefits for the providers of tourism services can be divided in international and domestic benefits, and equals the producer surplus obtained by these providers, i.e. the amount of income they receive minus the costs they incur for offering the service. With respect to the conservation of biodiversity, it is these domestic revenues that are most important, because they provide incentives for the maintenance of the biodiversity at the local and national levels. Table 14 presents the domestic revenues for tourism revenues in a number of African countries. It is clear that there are large differences between countries, with South Africa and Kenya benefiting most from international tourism. Key factors in attracting tourists appear to be wildlife (in particular the 'big five' species), tourist facilities and ease of access to parks, and the stability and safety of the country (MEA, 2006).

Whereas income provides a first indicator for the value of the tourism service, two other factors play a role in determining the economic value of the tourism service. First, the amount of money that seeps away from the tourism sector (e.g. because international hotel chains purchase a substantial part of food, linnen and other commodities on the international rather than the domestic market, and because their profits are transferred to owners or shareholders abroad). Second, the positive impact of tourism on other sectors (multiplier effect), e.g. through all kinds of local expenditures of staff working in hotels or restaurants.

Table 14.

Income from nature tourism in Africa.

	Non-African	African	Domestic	Non-African	African	Domestic	Total
Angola	0.8	0.1	..	0.3	0.0	..	0.3
Botswana	110.4	362.5	..	30.6	100.6	..	131.3
Burundi	0.0	0.0	..	0.0	0.0	..	0.0
Congo	0.0	0.0	..	0.0	0.0	..	0.0
Dem. Rep. Congo	0.0	0.0	..	0.0	0.0	..	0.0
Equatorial Guinea	0.0	0.0	..	0.0	0.0	..	0.0
Gabon	28.0	0.0	..	1.3	0.0	..	1.3
Kenya	552.8	201.6	0.2	178.2	65.0	7.5	250.7
Lesotho	5.4	48.4	..	0.6	4.9	..	5.5
Malawi	18.4	91.0	..	2.2	10.8	..	13.0
Mozambique	6.0	36.0	..	1.2	7.2	..	8.4
Namibia	96.6	263.4	..	45.3	202.3	..	247.6
Rwanda	1.7	0.0	..	2.7	0.0	..	2.7
South Africa	1,203.3	3,425.6	5.6	504.4	1,436.0	358.4	2,298.8
Swaziland	77.3	166.6	..	15.9	11.0	..	27.0
Tanzania	203.7	0.0	..	299.9	0.0	..	299.9
Uganda	28.0	92.8	..	27.6	91.6	..	119.2
Zambia	137.6	321.6	..	21.85	1.0	..	72.8
Zimbabwe	358.4	1,136.0	..	34.4	109.1	..	143.5
TOTAL	**2,828.4**	**6,145.6**		**1,166.4**	**2,089.5**		**3,622.0**

Source: MEA (2006)

TOOL 5 Assessing the costs & benefits of land use change

The fifth Tool involves a dynamic assessment of the impact of land degradation on the supply of ecosystem services, and the resulting economic damages. The Tool can also be used to analyze the benefits of enhancing or maintaining ecosystem services supply through SLM. The Tool depends on quantitative analysis of the relation between (i) degradation or SLM and ecosystem state; and (ii) ecosystem state and supply of ecosystem services. These two relations can be assessed through basic ecological economic modeling. As the analysis may only involve a very limited set of relations (e.g. erosion -> loss of fertile topsoil -> loss of crop production), they do often not require specific software or complex modelling skills. For instance, Excel can be used for basic quantitative analyses.

Two key steps in the ecological-economic assessment are (i) to link drivers to environmental state, and (ii) to link environmental state to ecosystem services (see Figure 8). The first steps requires analysis of which environmental compartments will be influenced by the drivers, and how these compartments will change following a change in the driver. The second step involves the linking of environmental compartments to ecosystem services. For instance, increased cropping intensity and reduced fallow periods in a slash-and-burn system reduce the fertility status of the soil which reduces the regrowth of vegetation and the supply of NTFPs. If the dose-response curve is known, the economic consequences of increased cropping intensity can be assessed.

SLM issues tend to be complex, involving a broad range of drivers and management alternatives, and not all drivers can be modeled and analyzed. However, often, it is possible to identify one or two key drivers, and to establish the relation between these drivers and ecosystem services supply. In these cases, it can be made clear which benefits can be obtained from changing land use practices. By means of an optimization approach, it is also be possible to identify optimal management strategies, i.e. the management strategy that generates the highest net benefits for society.

Purpose of the Tool

The purpose of this Tool is to guide the user in the application of ecosystem services modeling for SLM. It provides a basic structure for ecological-economic modeling of the relation between land use change and ecosystem services supply, as well as guidance on the application of the Tool through a case study.

How to use the Tool

The Tool provides a framework and general guidelines that can be used to model impacts of changes in ecosystems on the supply of ecosystem services. The tool allows the identification of principal drivers, their impact on the state of the ecosystem, and the resulting impacts on the supply of ecosystem services. In addition, in Step 5.3, it is examined how optimal ecosystem management strategies can be identified on the basis of ecological-economic modeling. As such a wide range of processes and ecosystem components can play a role, these guidelines can only provide the general approach, which will have to be fine-tuned for every individual impact assessment.

Step 5.1 Selection of relevant drivers and processes

Figure 8 presents a framework that can be used to identify the main drivers and processes that determine ecosystem change and subsequent changes in ecosystem services supply. The three main drivers relate to: (i) the (over-) harvesting of ecosystem services, specifically provisioning services; (ii) the impacts of pollutants on the system; and (iii) direct interventions in the ecosystem. This latter is a broad category that comprises such different interventions as construction of a road through an ecosystem or reforestation. The three drivers can cause changes in the state of the ecosystem, which are likely to affect the ecosystem's capacity to supply ecosystem services.

Note that the framework is fully compatible with the DPSIR framework and the Indicator Framework for SLM specified in the GEF Project 'Knowledge from the Land'. The economic system contains different drivers that may result in pressures on the ecosystem in the form of overharvesting of resources, pollution or other impacts. This leads to changes in the state of the ecosystem (or land use system), with, consequently, an impact on the supply of ecosystem services. Society may respond by reducing the pressures, or accept (or adapt to) lower levels of ecosystem services supply.

The framework presents a conceptual outline of an ecological-economic model or assessment. It is particularly suitable to assess the temporal scales related to ecosystem management. The framework, as presented in figure 8, does not distinguish between different spatial scales of ecosystem management. However, it can be adjusted in order to

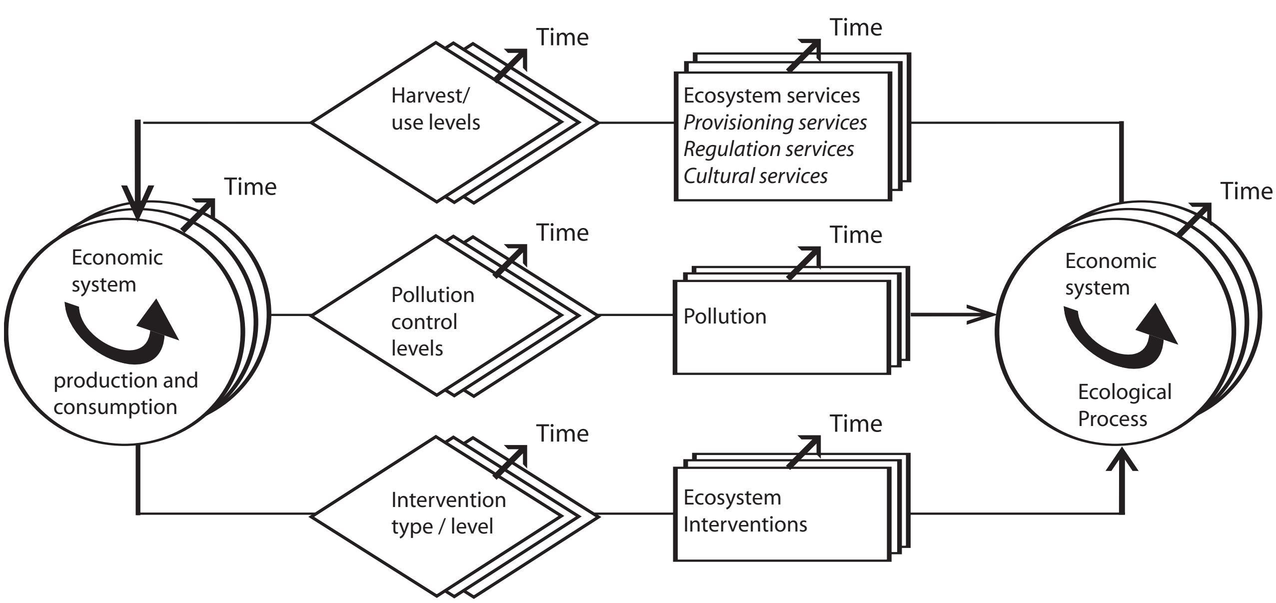

Figure 8. Conceptual framework for the ecological-economic modeling of the management options for a dynamic ecosystem. The square boxes are labels for the flows between the two systems, and the diamonds represent the decision variables.

become more spatially explicit by defining the relevant interactions at different spatial scales. This requires incorporation into a GIS system.

Application of the framework requires (i) identification of the relevant drivers and the potential management options; (ii) the modeling of ecosystem dynamics as a function of external drivers and internal processes; (iii) analysis of the costs of management options and valuation of the ecosystem services; and (iv) analysis of the economic efficiency and sustainability of management options. These four steps are briefly described below.

(i) Identification of the drivers and potential management options. This step first involves the identification of the interactions between the ecosystem and the economic system, including the current management of the system and the various services supplied by the ecosystem. Subsequently, the potential options to enhance the management should be identified. This may include, for instance, changing the harvest levels of ecosystem services or the application of pollution control technologies.

(ii) Modeling the dynamics of the ecological-economic system. This steps involves the modeling, in physical terms, of the impacts of management options on the ecosystem, and the impact of changes in ecosystem state on the system's capacity to supply ecosystem services. For systems subject to complex dynamics, it is important that these dynamics are reflected in the model. This requires the modeling of the main ecosystem components and the feedback mechanisms between them, including relevant non-linear and/or stochastic processes. In spite of the large number of ecological processes regulating the functioning of ecosystems, recent insights suggest that the main ecological structures are often primarily regulated by a small set of processes (Harris, 1999; Holling et al., 2002). This indicates that inclusion of a relatively small set of key components and processes in the model may be sufficient to accurately represent the (complex) dynamics of the system. This is further elaborated in Step 5.2 'Modeling changes in ecosystem services supply' below.

(iii) Analysis of the costs of management options and valuation of the ecosystem services. In this third step, the physical flows need to be expressed in a monetary measure. This involves both examining the costs of the management options, for example through the establishment of a pollution abatement cost curve, and the valuation of changes in the supply of ecosystem services following changes in management. Appropriate valuation methods differ per type of ecosystem service, as described in Tool 4 ('Economic Valuation').

(iv) Analysis of the impacts of ecosystem change and of the management options. Once the ecological-economic model has been constructed, it can be used to assess the impacts of the ecosystem change on the supply of ecosystem services, as well as of the efficiency and sustainability of different ecosystem management options. The efficiency of ecosystem management can be revealed through comparison of the net welfare generated by the ecosystem and the costs involved in maintaining and managing the ecosystem (e.g. Pearce and Turner, 1990). Through a simulation or algebraic optimization approach, efficient management options, i.e. management options that provide maximum utility given a certain utility function, can be identified. The sustainability of management options can be examined by analyzing their long-term consequences for the state of the ecosystem including its capacity to supply ecosystem services (Pearce et al., 1989; Barbier and Markandya, 1990). Optimization procedures are described in Step 5.3.

Step 5.2 Quantitative assessment of ecosystem change

Through dynamic systems modeling, the ecological-economic model can be specified. A systems modeling approach is based upon the modeling of a set of state (level) and flow (rate) variables in order to capture the state of the system, including relevant inputs, throughputs and outputs, over time. This may comprise a range of theoretical, statistical or methodological constructs, dependent upon the requirements and limitations of the model. The systems approach can contain non-linear dynamic processes, feedback mechanisms and control strategies, and can therefore deal in an integrated manner with economic-ecological realities (Costanza et al., 1993; Van den Bergh, 1996). Note that these models are not necessarily highly complex or require sophisticated software. Simple spreadsheet models capturing 2 to 4 key relations in equations may be sufficient for the quantitative assessment.

The modeler needs to identify the key drivers, state indicators and processes, and quantify the relations between them in terms of flow and state variables. For many ecosystem types, these are 'standard' models that indicate the general types of dynamics that can be expected as a function of drivers and pressures. Based on time-series data for the specific ecosystem involved, such general models can be calibrated in order to yield a realistic modeling of the system. Where more detailed modeling or analysis is required, specific software such as Stella is available, but simple spreadsheet programs may also be sufficient.

Step 5.3 Optimization of ecosystem management

Development of an ecological-economic model allows the user to calculate the impacts of a change in the ecosystem on the supply of ecosystem services. However, in addition, it allows the user to identify the optimal management approach to the ecosystem. For instance in a situation where pollution control costs money related to investment in waste treatment plants, and yields benefits in relation to an enhanced supply of ecosystem services from a clean lake, the ecological model allows selection of the most efficient pollution control level.

In the context of dynamic systems models, two approaches can be followed to determine the value of the decision variables that provides maximum utility: (i) a simulation (programming) approach and; (ii) an algebraic, static or dynamic optimization approach. In both cases an ecological–economic model is first developed, but the optimal solution is found in different manners. In the simulation approach, a model is developed to represent modifications in the ecosystem and the economic system, and the key interactions as a function of the decision variable(s). By simulating the development of the ecosystem for a range of values of the decision variables, optimal solutions can be revealed - within the tested range and under the tested conditions. In the algebraic optimization approach, optimal solutions are found in a numerical or algebraic manner, through the preparation of the Hamiltonian and solving the first and second order conditions (Chiang, 1992), as further discussed below.

According to the framework presented in figure 8, there are three principal types of ecosystem management: (i) changing the use level of ecosystem services; (ii) the control of pollution influxes; and (iii) direct interventions in the ecosystem. Below, it is analyzed how the efficiency of these three types of measures can be assessed, and which conditions need to be met to achieve efficient management.

(i) Optimizing the extraction of renewable resources. Efficient resource extraction has been studied since over a century. Early studies focused on forestry (Faustmann, 1849) whereas studies on fisheries management (e.g. Gordon, 1954) and grazing systems (e.g. Dillon and Burley, 1961) are more recent. The standard models assume a logistic growth curve, with low resource growth at low population sizes and at population sizes close to the carrying capacity. In addition, these models may consider quality and price changes, cost for inputs and harvesting costs. Forest management models have dealt with, in particular, the choice of the optimal rotation period, while in fisheries and grazing systems, the key decision variable is the harvest rate.

In a deterministic, dynamic, single species model, the efficient stock and harvest level depend upon the marginal growth rate of the stock, and the discount rate used (e.g. Tietenberg, 2000). The stock's marginal growth rate determines the rents that can be obtained from the natural capital stock, whereas the discount rate indicates the rents that can be obtained from depletion of the natural capital stock and investing the benefits in man-made capital. For instance, Clark (1976) assumed fixed harvest costs (i.e. harvest costs independent from the stock size) and showed that if the reproduction rate of the resource is lower than the discount rate, it may be efficient, from a utilitarian point of view, to harvest the full stock. This situation does not generally apply, as normally the harvest costs will increase with decreasing stock levels. Moreover, there may be a range of hidden costs related to overharvesting of particular species through the disturbance of the ecosystem, which may affect the whole range of ecosystem services supplied by the ecosystem (Jackson et al., 2001).

(ii) Efficient levels of nutrient pollution control. Land degradation may lead to run-off of fertilisers or sediments into rivers and other waterways. The optimal level of nutrient pollution or sedimentation is usually discussed in terms of the intersection of the marginal damage function and the marginal control cost function (see e.g. Tietenberg, 2000). The marginal damage function shows the damage resulting from pollution as a function of emissions of a particular pollutant. The marginal control cost function shows the cost of reducing emissions of the pollutant below the level that would occur in an unregulated market economy. The marginal damage function is composed of a chain of functional relationships, as depicted in Figure 9. Dispersion processes and chemical transformations may reduce local pollution loads. In some types of ecosystems, time lags may play a role, for example if there are buffers in the ecosystem that absorb pollutants, and release them once the input of pollutants has decreased (Carpenter et al., 1999).

Ecological-economic modeling of pollution control requires analysis of four main elements: (i) the costs of pollution control; (ii) the relation between dispersal of pollution and the build-up of pollution loads in the ecosystem; (iii) the impact of pollution loads on the capacity of the ecosystem to provide goods and services; and (iv) the benefits foregone as a result of a loss of ecosystem services.

(iii) Optimization of ecosystem intervention. In view of the diversity of possible ecosystem interventions, the efficient level of ecosystem intervention can only be analyzed in general terms in this section. If the evaluation concerns only one, discrete measure, the basic criterion in terms of efficiency is whether the discounted benefits of the measure exceed the discounted costs of the measure, or not. The benefits include the potential impact of the measure on the supply of all relevant ecosystem services, and the costs include investment costs, operation and maintenance costs, and possible negative impacts on the supply of other ecosystem services (Hanley and Spash, 1993).

In case a range of measures is possible, the efficient intervention level corresponds to implementation of those measures that minimize the sum of the total costs of the measures and the costs resulting from a loss of environmental quality (see e.g. Hanley and Spash, 1993). A loss of environmental quality may cause a loss of ecosystem services, and bring costs for compensation payments to stakeholders impacted by that loss (Hueting, 1980). For concave benefit and convex cost functions, the marginal benefits of implementing the measure equal the marginal costs of adverse environmental quality at the point of maximum efficiency (Tietenberg, 2000).

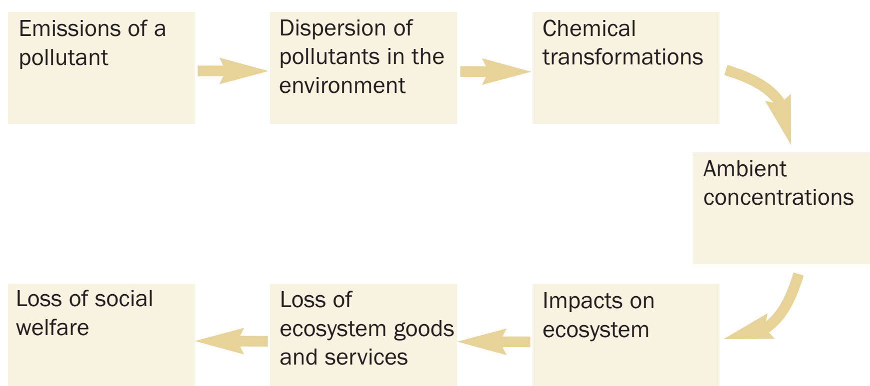

Figure 9. Schematic overview of a marginal damage function

Case study 6. Costs of soil nutrient depletion in Southern Mali

This case study (derived from Van der Pol and Traore, 1993) demonstrates how the costs of soil nutrient depletion can be calculated with respect to agricultural production in Mali. First, the degree of soil mining by agricultural production is assessed by calculating nutrient balances: differences between the amount of plant nutrients exported from the cultivated fields, and those added to the fields. Second, the costs of nutrient depletion are calculated.

Nutrient export processes include extraction by crops, losses due to leaching, to erosion, and to volatilization and denitrification. Inputs include applications of fertilizer and manure, restitution of crop residues, nitrogen fixation, atmospheric deposition of nutrients in rain and dust, and enrichment by weathering of soil minerals. Nutrient balances are calculated for N, P, K, Ca, and Mg. The resulting figures indicate large deficits for nitrogen, potassium and magnesium. For the region as a whole, the calculated annual deficits are -25 kg N/ha, -20 kg K/ha, and -5 kg Mg/ha. Further, acidification is to be expected, in particular in areas where cotton is grown. The deficits are caused by traditional cereal crops, but also by cotton and especially by groundnut. The latter two crops are fertilized, but insufficiently. For phosphorus and calcium the balance of the region as a whole appears to be about in equilibrium, but locally large variations may occur. Furthermore, erosion and denitrification are important causes of nutrient loss, accounting respectively for 17 and 22% of total nitrogen exports. Atmospheric deposition and weathering of minerals in the soil are still important nutrient inputs that contribute as much nutrients as organic and mineral fertilizer combined. Hence, nutrient depletion is very large in comparison to the amount of fertilizer applied. Drastic options, such as doubling the application of fertilizer or manure, or halving erosion losses, even if feasible, would still not be enough to make up for the calculated deficits.

In the second step, the costs of nutrient depletion are calculated. This is done following a basic Replacement Cost method (one of the Averting Behavior Methods described in Section 4.2.1). The economic value of nutrient deficits and surpluses has been calculated on the basis of prices which farmers had to pay for fertilizers (financial prices) in 1989. In line with the preceding physical considerations, the economic evaluation shows a considerable contribution of soil nutrient depletion to farmer's income. Average nutrient deficit per hectare values around FCFA 15,000 (US$ 62/ha) in 1989. Compared to income from agricultural activities this represents a substantial proportion. Evaluating all harvested products, including cereals, at 1989 market prices, the average gross margin from agricultural activities in the study region amounts to FCFA 34,200/ha (US$ 134/ha). Thus, in 1989, soil nutrient depletion represented as much as 44% of the average farmer's income (Van der Pol and Traore, 1993).

Case study 7. Efficient nutrient pollution control in the De Wieden wetlands

This case study demonstrates how ecosystem services valuation can be applied in a dynamic context in order to define optimal ecosystem management options. It is based on a recent, more elaborate study by Hein (2006) on water management in the 'De Wieden' wetlands in the Netherlands. The case study compares the costs and benefits of eutrophication control measures, given that eutrophication control leads to enhanced water quality and an enhanced supply of ecosystem services by the lakes of De Wieden. Eutrophication of the lakes is caused by agricultural run-off from exces fertilizer use in surrounding farmland.

The study considers water quality in four lakes with a total area of 1640 ha and an average water depth of only 1.8 meter. The ecosystem services provided by the lakes are nature conservation and recreation, reed cutting and fisheries. For the study, an ecological-economic model was developed that describes the response of the ecosystem to eutrophication control measures. Total-P concentrations are used as the control variable of the models because P is the main limiting nutrient in the lakes. The various steps included in the model are presented in figure 10. The benefits of the transition to clear water are expressed as net present value (NPV) in order to compare them with the costs of eutrophication control measures.

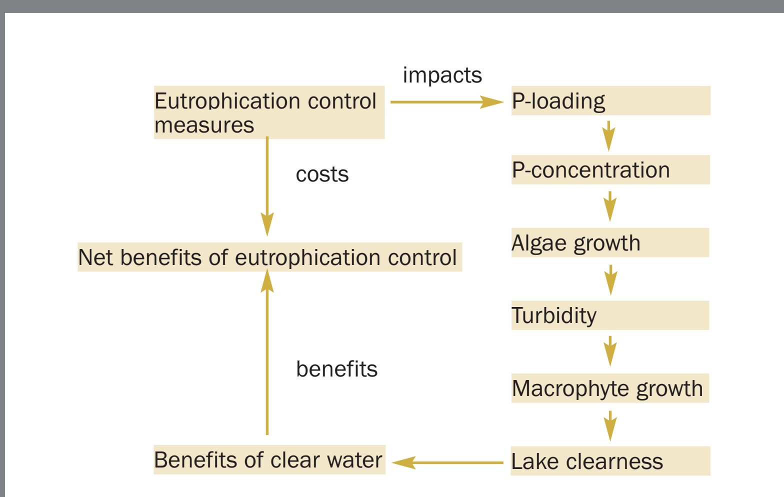

Figure 10. Model lay out (see text for explanation).

The three main steps of the model deal with: (i) the costs and impacts of mitigation measures; (ii) the modeling of the response of the ecosystem to eutrophication control measures; and (iii) analysis of the benefits of clear water. Subsequently, (iv) the costs and benefits of the measures can be compared. These steps are described below.

(i) Costs and impacts of mitigation measures. Potential measures available to reduce the inflow of phosphorus in the De Wieden wetland have been examined by the local waterboard. In collaboration with the main stakeholders in the area (nature conservationists, farmers, representatives from the tourist sector), they have identified the most feasible measures in terms of cost-effectiveness and acceptance for local stakeholders. This includes such measures as enhanced sewage treatment facilities or enhanced connection of remote houses to the sewage system.

(ii) Modeling the response of the ecosystem to eutrophication control measures. The model analyses how a reduction P-loading leads to a reduction in the steady state concentration of P in the lakes, and how this changes algae growth and, subsequently, turbidity, macrophyte water plant growth and the parts of the lake with clear water. The model also contains a threshold in lake water turbidity; in line with ecological models of shallow lake dynamics (Scheffer, 1998), it is assumed that at a certain threshold water plants will start growing, reducing sediment resuspension, providing a habitat for Daphnia (waterflees) and bringing changes in the fish community (from bream to a more diverse community dominated by pike). The model contains six formula that capture the response o the ecosystem to nutrient

Case study 7. (cont.)

loading. They deal with phophorus inflow, algae growth, chlorophyl contents and water transparency. The have been constructed on the basis of ecological theory, and calibrated with local data on nutrient loading and water quality. These guidelines can not present the formulas themselves, for which the reader is referred to Hein (2006).

(iii) Analyzing the benefits of a switch to clear water. Only the two services nature conservation and recreation will benefit from a switch to clear water in De Wieden. Regarding nature conservation, a range of threatened species is expected to benefit from a switch to clear water, and there are no rare or threatened species that would decline from such a shift. As for the recreation service, especially swimmers but also sailors and surfers appreciate clear water, provided that waterplants do not hamper the access of the boats to the lakes. Fisheries and reed cutting will probably not significantly benefit from a transition to clear water. For local fisheries, the most important species is eel, which is relatively insensitive to modest changes in P concentrations or a potential shift to clear water. Reed growth also does not respond to such changes.

The monetary benefits of a switch to clear water resulting from increased tourism and nature conservation are difficult to quantify. Therefore, the model calculates the net benefits of a reduction in total-P loading for a range of assumed values of the increased supply of the nature conservation and recreation service following a switch to clear water. In other words, the net benefits of eutrophication control measures are calculated as a function of both (i) the level of eutrophication control and the type of measures implemented (without or with biomanipulation); and (ii) the assumed value of the marginal increase in the supply of the two ecosystem services.

(iv) Comparison of the costs and benefits of eutrophication control measures. Figure 11 shows the economic efficiency of reducing the inflow of P in De Wieden for benefits of enhanced recreation and nature conservation valued at 1 million euro per year. There is a bimodal distribution; there is a local maximum efficiency at zero reduction in P-loading, and a – higher – maximum for a reduction in P-loading of 2 ton/year. The second local maximum corresponds to the minimum P inflow reduction at which the complete lake changes from the current turbid water state to a clear water state. This P concentration is 0.09 mg/l. The model shows that if the annual marginal benefits provided by clear water (through enhanced biodiversity protection and better opportunities for recreation) are valued at at least 0.2 million euro, it is economically efficient to reduce the inflow of total-P with 2 ton/year in order to obtain clear water. The model presented in this case study is now being used by the local Waterboard to analyze the economic impacts of eutrophication control measures (Hein, 2006).

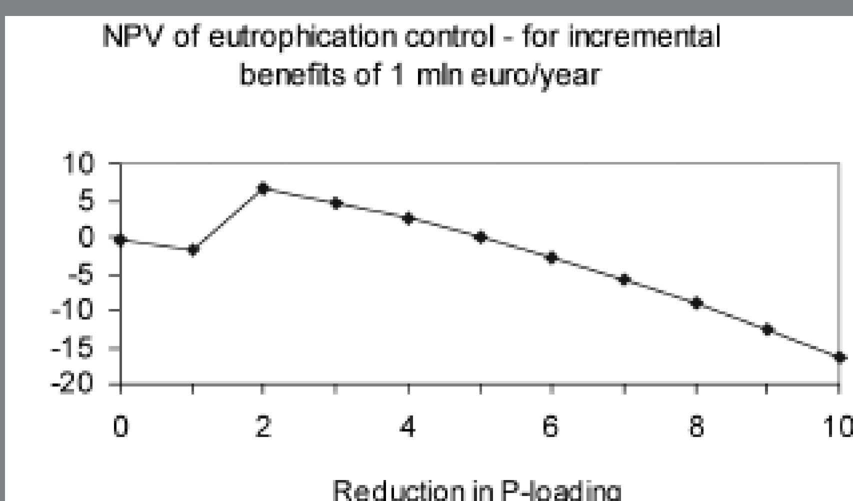

Figure 11. Net benefits of reducing P loading.

Case study 8. Costs of coral bleaching in the Indian Ocean

As a consequence of higher sea water temperatures due to global warming and anomalies in El Niño events (which may also be related to global warming), the last decade has witnessed unprecedented coral bleaching world-wide. Coral reefs are highly vulnerable to changes in water temperature, which may induce them to excrete their symbiotic algae. The corals change color (the "bleaching") and will die after several weeks to months. In the Indian Ocean, coral bleaching was particularly severe in 1997 and 1998, which were among the warmest years on record and which witnessed strong El Niño events. In some parts of the Indian Ocean, water temperatures rose as much as 4 oC above the long term average (Westmacott et al., 2001).

The warm seawater temperatures in the Indian Ocean had a devastating impact on coral reefs throughout the region. Mortality rates went up to 95% in parts of India, Sri Lanka, Maldives, Kenya, Tanzania and the Seychelles. Most bleaching occurred in water < 15 meter depth but, unlike most other bleaching events, this time also corals in up to 50 meter deep water were heavily affected. Furthermore, not only fast growing coral species, but all species of coral were affected (Wilkinson et al., 1999).

The bleaching and subsequent dying of corals has a number of important ecological impacts. A dead coral is likely to break down and form a bed of rubble within a few years. Besides the corals themselves, large numbers of fish, invertebrate and plant species will be affected. This also brings changes in the ecological dynamics of the system. In case a major part of the coral is damaged, algae may start occupying space on the reef, preventing the return of coral. There are also significant economic effects. Mass coral death may affect local fisheries, tourism and coastal protection. These impacts and costs are summarized below, with respect to three main ecosystem services provided by coral reefs: (i) fisheries, (ii) tourism, (iii) storm protection. Note that coral reefs may provide a range of other products and services as well, such as the supply of aquarium fish, medicinal compounds, etc. (Moberg and Folke, 1999).

Impacts on fisheries. Reefs generate a variety of seafood products such as fish, mussels, crustaceans, sea cucumbers and seaweeds. Reef-related fisheries constitute approximately 9–12% of the world's total fisheries, and in some parts of the Indo-Pacific region, the reef fishery constitutes up to 25% of the total fish catch (Cesar, 1996). In the short term, few impacts of coral bleaching on local fisheries were noted in two case studies in Kenya and Tanzania (Westmacott et al., 2001). After a year, dead corals were still standing and there was no significant change in commercial fish species, except for a small increase in herbivorous fish that benefited from more abundant algae. However, in the longer term, bleaching significantly reduces the productivity of coral reefs, in particular once the reefs physically collapse. An indication of the potential costs of a loss of fisheries after such a collapse of the coral is provided by Cesar (1996), who estimates that large scale physical damage (due to coral mining) in Indonesia causes economic losses due to lost fishing opportunities in the order of $94,000 per year per sq km of reef.

Impacts on tourism. For many countries around the Indian Ocean, tourism is a key economic asset. Tourism is the biggest sector in many of the small island states in the region including the Maldives, Mauritius, the Comores and the Seychelles. For many tourists, diving and snorkeling are among the main reasons to visit these island states (e.g. in the Maldives, 45% of all tourists dive). Westmacott et al. (2001) showed that coral bleaching reduces the interest of visitors in diving, and also reduces the interests of tourists to visit the islands. Hence, coral bleaching reduces both the expenditure of tourists and the number of tourist arrivals. The economic damage costs of coral bleaching due to a loss of tourism are shown, for selected islands, in Table 15. The losses represent the damage costs for the islands only, the loss of welfare for tourists is not included in the figures.

Case study 8. (cont.)

Costs of a loss of storm protection. The importance of coral reefs for coastal protection depend on a range of factors, including the proximity of the corals to the coast, the size and depth of the coral reefs and the presence or absence of other protective systems (such as mangroves). Hence, there is large variation in the importance of this ecosystem service supplied by coral reefs. The following studies present an indicative estimate of the storm protection value of coral reefs. Berg et al. (1998) use the cost of land loss as a proxy for the annual cost of coastal erosion due to coral mining in Sri Lanka. Depending on land price and use, these costs are between US$ 160,000 and US$ 172,000 per km of reef per year. Cesar (1996) uses a combination of the value of agricultural land, costs of coastal infrastructure and houses to arrive at value for the storm protection service in Indonesia of between US$ 90,000 and US$ 110,000 per km of reef per year.

Table 15.

Economic costs of tourism losses due to coral bleaching

Area / Country	Economic damage from loss of tourism due to coral bleaching (US$ million per year)
Zanzibar	4
Mombassa	17
Maldives	3

Source: Westmacott et al. (2001)

Conclusions

The Toolkit presents three main approaches that can be followed to analyze and value the economic costs of land degradation and the benefits of sustainable land management. These approaches are: (i) partial valuation; (ii) total valuation; and (iii) impact assessment. Partial valuation can be used to analyze the importance of ecosystems, or the benefits of sustainable management, in relation to the provision of a limited set of ecosystem services. Total valuation involves valuing all services provided by an ecosystem, and can be used, for instance, to compare the costs and benefits of different types of land use options (e.g. sustainable versus non-sustainable land use). Impact assessment is a more dynamic approach that allows analyzing the economic impacts of gradual changes in land management, for instance because of the adoption of SLM.

In general, there is a high potential to use ecosystem services valuation to support promotion of SLM in developing countries. In a developing country setting, there are often less financial resources to spend on conservation of natural resources for the sake of the natural resource in itself. At the same time, there is a high dependency of the national population on natural resources, in particular in those countries where agriculture and other resources provide the main source of income for large parts of the population. In these circumstances, sustainable provision of ecosystem services often makes an important to the income and livelihood of local people, even though these benefits are not always fully reflected in market transactions. Economic valuation of these services allows therefore obtaining a proper understanding of these various benefits, and of the need to consider them in land and environmental management decisions in order to maintain welfare of the local population.

Payment for Ecosystem Services (PES) schemes can make an additional contribution to environmental management, although caution needs to be taken as their effectiveness strongly depends on the socio-economic setting involved. For instance, poverty may restrict the effectiveness of PES. Poor stakeholders can not be expected to start paying for ecosystem services they earlier received for free. In addition, transaction costs may be high, as there is a need to monitor the supply of the ecosystem service over time, and as a trustworthy mechanism has to be set up to organize the transfer of payments among stakeholders.

A number of general recommendations can be provided for the economic analysis of land degradation and SLM. First, the objective of the study needs to be clear, as the objective determines the scale and the system boundaries, the appropriate valuation methods, and the data requirements. Second, care needs to be taken to analyze both the ecological and the economic aspects of the ecosystem services involved. In particular for the regulation services, it is often as time-consuming to quantify the service in ecological or biophysical terms (Tool 3) as it is to conduct the actual valuation itself (Tool 4). Third, the uncertainties in the analysis need to be discussed, the impact of the study will depend on the amount of credit it will obtain and it is important to communicate how reliable the study's outcomes are. Fourth, valuation studies require an interdisciplinary approach involving economists, ecologists, hydrologist, sociologists, etc., depending on the functions and environmental setting to be studied.

In view of the general importance of economic arguments in decision making on land use, it is anticipated that this Toolkit can support the design and implementation of land use policies. Often, there is a much smaller difference between economic efficient and sustainable land use than generally perceived.

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Appendix 1. Glossary

Agro-ecosystem: a dynamic association of crops, pastures, livestock, other flora and fauna, atmosphere, soils, and water. Agroecosystems are contained within larger landscapes that include uncultivated land, drainage networks, rural communities, and wildlife.

Bequest Value: the value that people derive from knowing that their off spring will be able to benefit from an ecosystem service.

Biomass: the total weight of a designated group of organisms in a particular area.

Consumer surplus: the difference between the price actually paid for a good, and the maximum amount that an individual is willing to pay for it. For instance, if a person is willing to pay up to $10 for something, but the market price is $4, then the consumer surplus for that item is $6.

Compensating variation: the amount of money that leaves a person as well off as he was before a change. Thus, it measures the amount of money required to maintain a person's satisfaction, or economic welfare, at the level it was at before the change.

Contingent Valuation: a formal survey technique that requires respondents to specify their preferences for different goods or services and how much they would pay to obtain them.

Cost-benefit analysis: a comparison of economic benefits and costs to society of a policy, program, or action.

Cultural services: the benefits people obtain from ecosystems through recreation, cognitive development, relaxation, and spiritual reflection. In this Toolkit, the habitat service (i.e. the benefits people derive from the protection of biodiversity and nature for the sake of nature itself) is also included as a cultural service.

Demand curve: the graphical representation of the demand function. The demand curve indicates how many units of a good will be purchased at a certain price. In general, at higher prices, less will be purchased, so the demand curve slopes downward. The market demand function is calculated by adding up all of the individual consumers' demand functions.

Discount rate: the rate used to reduce future benefits and costs to their present time equivalent.

Economic efficiency: the allocation of goods or services to their highest relative economic value.

Ecological processes: the physical, chemical, and biological processes that maintain and support the functioning of the ecosystem. Examples of ecosystem processes are denitrification, primary production, and evapotranspiration.

Ecosystem approach: the ecosystem approach is a strategy for the integrated management of land, water and living resources that promotes conservation and sustainable use in an equitable way. An ecosystem approach is based on the application of appropriate scientific methodologies focused on levels of biological organization, which encompass the essential structure, processes, functions and interactions among organisms and their environment.

Ecosystem function: the capacity of an ecosystem to provide goods and services that satisfy human needs, directly or indirectly. Ecosystem functions depend upon the state and the functioning of the ecosystem. For instance, the function 'production of firewood' is based on a range of ecological processes involving the growth of plants and trees that use solar energy to convert water, plant nutrients and CO_2 to biomass.

Ecosystem services: the goods or services provided by the ecosystem to society. In order for an ecosystem to provide services to humans, some interaction with, or demand from, people for the good or service concerned is required.

Equivalent variation: the amount of money that leaves a person as well off as they would be after a change. Thus, it measures the amount of money required to maintain a person's satisfaction, or economic welfare, at the level it would be at after a change.

Eutrophication: The process by which a body of water accumulates nutrients, particularly nitrates and phosphates. This process can be accelerated by nutrient-rich runoff or seepage from agricultural land or from sewage outfalls, leading to rapid and excessive growth of algae and aquatic plants and undesirable changes in water quality.

Existence value: the value that people place on knowing that something exists, even if they will never see it or use it.

Externalities: uncompensated side effects of human actions. For example, if a stream is polluted by runoff from agricultural land, the people downstream experience a negative externality.

Fixed costs: production costs that are not related to the level of production; also referred to as overhead costs.

Geographical Information System (GIS): a computer mapping system that links databases of geographically-based information to maps that display the information.

Habitat: The place where a population of plants or animals and its surroundings are located, including both living and non-living components.

LDC: Least Developed Country

Market failure: the inability of markets to reflect the full social costs or benefits of a good, service, or state of the world. Therefore, markets will not result in the most efficient or beneficial allocation of resources.

Net economic benefit: the net economic benefit is the total economic benefit received from a change in the state of a good or service, measured by the sum of consumer surplus plus producer surplus, less any costs associated with the change.

Net Present Value (NPV): the sum of the present and discounted future flows of net benefits (expressed as e.g. US$/ha). A discount rate is used to reduce future benefits and costs to their present time equivalent.

Non-use values: values that are not associated with actual use, or even the option to use a good or service.

Opportunity cost: the value of the best alternative to a given choice, or the value of resources in their next best use. For instance, the opportunity costs of a natural park may be the value that could be derived from converting the area to agricultural land use. The opportunity costs may be higher or lower than the value of the resource under present management.

Option value: the value that people place on having the option to enjoy something in the future, although they may not currently use it.

Producer surplus: the difference between the total amount earned from a good (price times quantity sold) and the production costs.

Provisioning services: the goods and services extracted from an ecosystem, either through harvesting (collecting a piece of fruit in a forest), or through extensive or intensive agriculture. Valuation of these services always needs to consider the amount of effort (i.e. costs) required to obtain or produce the good or service.

Public goods: In the case of public goods, the availability of a good to one individual does not reduce its availability to others (non-rivalry) and the supplier of the good cannot exclude anybody from consuming it (non-excludability). For example, safety provided by dykes is a pure public good.

Regression analysis: a statistical process for fitting a line through a set of data points. It gives the intercept and slope(s) of the "best fitting" line. Thus it tells how much one variable (the dependent variable) will change when other variables (the independent, or explanatory, variables) change.

Regulation services: services provided by the ecosystem involving the regulation of climate, hydrological and bio-chemical cycles, earth surface processes, and biological processes.

Renewable resource: a resource that is capable of being replenished through natural processes (e.g., the hydrological cycle) or its own reproduction, generally within a time-span that does not exceed a few decades. Technically, metal-bearing ores are not renewable, although metals themselves can be recycled.

Shadow price: price adjusted to eliminate any distortions caused by politics or market imperfections in order to reflect the true willingness to pay.

SIDS: Small Island Developing State

Substitute goods: goods that you might purchase instead of a particular good. For example, different types of bread are substitutes for each other.

Supply Function : the mathematical function that relates price and quantity supplied for goods or services. The supply function tells how many units of a good that producers are willing to produce and sell at a given price.

Supply Curve: the graphical representation of the supply function. Because producers would like to sell more at higher prices, the supply function slopes upward.

Sustainable development: development that meets the needs of the present without compromising the ability of future generations to meet their own needs

Total economic value: the sum of the direct use, indirect use, option and non-use values for a good or service.

Threshold: when used in reference to a species, an ecosystem, or another natural system, it refers to the level beyond which further deterioration is likely to precipitate a sudden adverse, and possibly irreversible, change.

Use value: value derived from actual use of a good or service. Uses may include indirect uses. For example, the buffering impact of upstream forests on downstream water flows provides an indirect use value of the forest for downstream water users.

Variable costs: production costs that change when the level of production changes, so that when more is produced the costs increase; as opposed to fixed costs.

Wetlands: lands where water saturation is the dominant factor in determining the nature of soil development and the types of plant and animal communities

Willingness to Pay: the amount of money (or goods or services) that a person is willing to give up to obtain a particular good or service.

Appendix 2. Useful references on the internet

Association of Environmental and Resource Economists http://www.aere.org/

Ecosystem Services Project http://www.ecosystemservicesproject.org Various publications on methods and applications in the field of ecosystem services valuation, with a focus on Australia.

Ecosystem Valuation Website : www.ecosystemvaluation.org Detailed information on valuation methods.

ENVALUE: http://www.epa.nsw.gov.au/envalue/ A searchable, global environmental valuation database developed by the NSW EPA (Australia). Systematic collection of environmental valuation studies presented in an on-line database. Summaries and results reported in the database were subject to a process of peer review.

Environmental Valuation & Cost-Benefit News : http://envirovaluation.org/ Empirical cost-benefit and environmental value estimates

Environmental Valuation Reference Inventory (EVRI) http://www.evri.ec.gc.ca/evri/
Searchable storehouse of over 800 empirical studies on the economic value of environmental benefits. Information in the EVRI is available to subscribers only

International Society for Ecological Economics http://www.ecologicaleconomics.org/

OECD : www.oecd.org. Statistical information plus technical reports in the field of environmental economics and sustainable development.

Resources for the Future http://www.rff.org/ Home page for Resources for the Future, a nonprofit organization that conducts research on environmental and natural resource issues

World Bank Environmental Valuation http://wbln0018.worldbank.org/environment/EEI.nsf/all/ General overview of economic valuation of environmental impacts including detailed information and links to case studies and applications.