TECHNICAL BRIEF

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POLICYMIX - Assessing the role of economic instruments in policy mixes for biodiversity conservation and ecosystem services provision

Guidelines for biodiversity valuation and benefits assessment of economic instruments

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POLICYMIX WP 4 Guidelines for biodiversity valuation and benefits assessment of economic instruments

Abstract

These guidelines set out a general framework for the valuation of biodiversity and ecosystem services in relation to economic policy instruments aiming for biodiversity conservation and ecosystem services provision. The steps for assessing the economic impacts of policy instruments are outlined, followed by the economic valuation of biodiversity and ecosystem services.

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1 Contents

1 Introduction

This Technical Brief aims to provide a concise set of guidelines to estimate the economic value of employing economic instruments as part of a policy mix for biodiversity conservation and ecosystem service provision. The main challenge here is twofold. First, to identify and isolate the welfare impacts of an economic instrument in a mix of policy instruments, and second to relate the impact on biodiversity and the ecosystem services involved. A considerable literature exists focusing on the economic value of biodiversity conservation. The value added of this report is found in the assessment of the economic value of biodiversity conservation directly linked to the use of economic instruments and their impacts on ecosystem services. In this context, valuation methods are used principally for the evaluation of instrument characteristics, and in second place for valuation since by varying the institutional framing the value is expected to vary. The report differs in this way from the recently published The Economics of Ecosystems and Biodiversity (TEEB) book and reports[1], which focus primarily on ecosystems and their valuation.

The welfare effects of economic policy instruments on ecosystem services provision through biodiversity conservation can be measured and valued in market and non-market terms. A large number of manuals and handbooks exist regarding the economic valuation of environmental change, more recently referred to as ecosystem goods and services. This Brief will not try to repeat what is already out there, at most an overview will be given of useful existing manuals (links to references are provided in the text). Of prime interest here is the assessment of the value added created by the use of economic instruments in biodiversity conservation and related ecosystem service provision. Preliminary reviews of the impact of PES schemes on ecosystem service provision and associated welfare changes, for instance in Latin America where a large number of existing PES schemes are found, are very critical, especially regarding the incremental impact of PES on ongoing biodiversity conservation efforts.

This Brief is structured as follows. Section 1 presents a conceptual framework for the valuation of the economic impacts of economic instruments for biodiversity conservation, more specifically based on the concept of payments for ecosystem services. Section 3 describes the general steps that have to be taken to evaluate the economic impacts of using economic instruments. Section 4 deals with the costs that are related to the use of policy instruments. Section 5 discusses the question of how to value the benefits of biodiversity conservation. This section includes, among others, a critical reflection on existing (meta-analyses of) valuation studies. The Brief draws conclusions based on experiences in the case studies in Section 6.

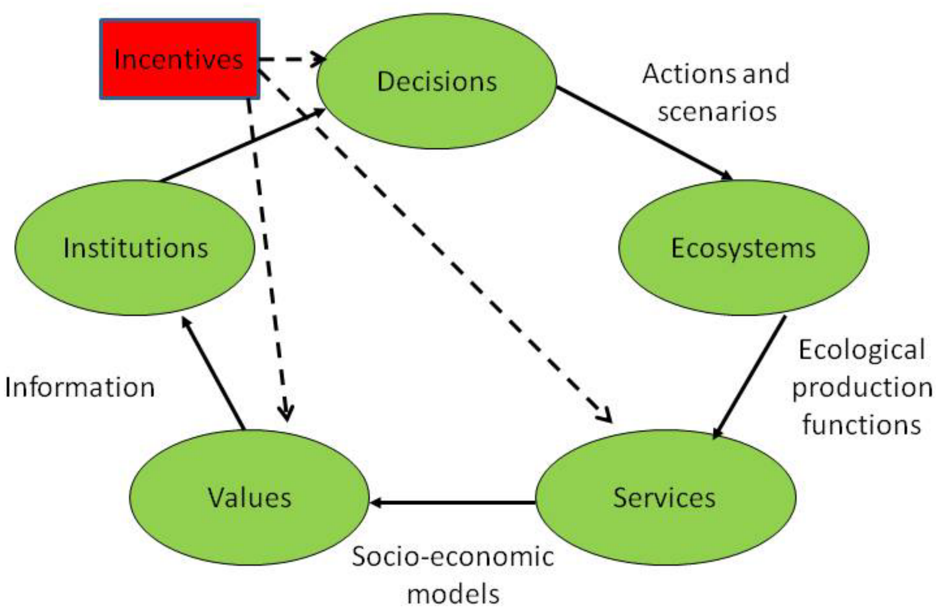
[1] See http://www.teebweb.org/.

2 Valuation framework

The overall valuation framework is presented in Figure 1. The framework applies to both ex ante and ex post evaluation of biodiversity conservation policy. Figure 1 illustrates on the top right-side that human behavior, be it previous actions from individual households, companies in the private sector or future government policy scenarios, impacts on our natural environment and the functioning of ecosystems. These impacts include the effects of land clearing and habitat modification, changes in species populations from harvesting activities (hunting and fishing), changes in nutrient flows from fertilizer application and runoff, changes in the hydrological cycle from water withdrawals and operation of dams, changes in local air and water quality from discharge of pollutants, and changes in global climate from emissions of greenhouse gases.

The guidelines address mainly how to quantify “values” in Figure 1; how to quantify the costs and benefits resulting from the application of a policy such as implementation of incentives, including the costs and benefits of decisions made by land users.

Figure 1: Integrative framework for the evaluation of payments for ecosystem services



Source: adapted from Daily et al. (2009).

Guidelines have also been produced on other aspects of Figure 1, and are downloadable from the POLICYMIX webpage:

- Ecosystems: Policy outcomes: A guideline to assess biodiversity conservation and ESS provision gains(WP3)
- Decisions and Values: Guidelines for Assessing Social Impacts and Legitimacy in Conservation (WP5)
- Institutions: Guidelines for the analysis of institutions shaping biodiversity policy instrument applicability (WP6)

A fundamental element of the ecosystem services paradigm is the recognition that changes in ecosystem structure or function influence the provision of ecosystem services. Ecological production functions can be used to understand how various ecosystem services are produced and how changes in ecosystem conditions affect the provision of these services. The basic understanding required for developing ecological production functions comes from ecology and other natural sciences. Ecosystem services in turn contribute to human welfare. A fundamental principle of economics is that these contributions can be represented as the benefits of an increase in the flow of ecosystem services or the cost of a decrease in flows, where benefits and costs reflect the preferences of the individual stakeholders affected by the change. The value of the change in the flow of an ecosystem service, as defined in economics, is measured in terms of the trade-offs that those individual stakeholders are willing to make, regardless of their underlying motivations. Both market and nonmarket valuation methods can be used to estimate these trade-offs based on socio-economic models developed for value elicitation and estimation.

Information about the benefits and costs of changes in the flow of ecosystem services can then be used to assess the net benefits associated with alternative policy options or outcomes. Although few economists believe that information about the net benefits of alternatives should be the sole basis for social choice, nearly all believe that it should be an important consideration in public policy decisions. The information about the socio-economic values associated with the change in the flow of ecosystem services feeds into institutions where the available information is converted into more or less adequate and effective policy incentives aimed at rewarding good behavior that helps to conserve biodiversity or punishing bad behavior that contributes to the degradation and destruction of biodiversity. This includes public policy decisions, which create incentives that affect the private decisions by firms and individuals, which in turn result in actions that affect ecosystems.

The translation of information about the costs and benefits of alternative courses of action and their trade-offs into financial-economic incentives is of most interest in this Brief. The incentives are considered to be effective if they contribute significantly to reaching the identified environmental objectives. The incentives are meant to change decision-making and corresponding behavior. In the economic analysis, the role and impact of the investigated financial-economic incentive in a mix of policy instruments will be measured in terms of the induced final outcome(s) and their welfare implications. This implies measuring the incremental change in ecosystem service provision and associated values (the dotted lines from incentives to ecosystem services and values).

3 General steps to evaluate the economic impacts of policy instruments

A number of general steps can be identified to assess, quantify and value the economic value(s) of the impact(s) of an incentive on the change in provision of ecosystem services and associated economic values (trade-offs). These steps are presented in Box 1 below.

Box 1: General steps to evaluate the economic welfare implications of the introduction of an economic incentive

Step 1: Define the incentive involved, including its key characteristics such as environmental objective, implementing agent, target group(s), geographical boundaries and time horizon, unit price, payment frequency, associated monitoring system etc.

Step 2: Define the baseline mix of policy instruments, including institutional-economic context, in which the incentive is or will be introduced (see WP6 guidelines).

Step 3: Determine the outcome of the situation without the incentive involved in terms of level of biodiversity and associated ecosystem service provision and economic welfare implications, either through a before-after comparison or matching procedure for the purpose of cross-section comparison ('*control treatment*') (see WP3 guidelines regarding matching).

Step 4: Determine the outcome of the situation with the incentive involved in terms of level of biodiversity and associated ecosystem service provision and economic welfare implications ('*experimental treatment*').

Step 5: Estimation of the additional costs of implementation and the transaction costs of the incentive involved (see sections 3 and 4).

Step 6: Estimation and valuation of the relevant welfare effects with and without introduction of the incentive involved using market and non-market valuation methods (see section 5).

Step 7: Comparison of the relevant welfare effects and estimation of the net benefits of the incentive over the relevant geographical and temporal scale.

Step 8: Sensitivity analysis of key uncertainties and assumptions underlying the estimation of the relevant welfare effects.

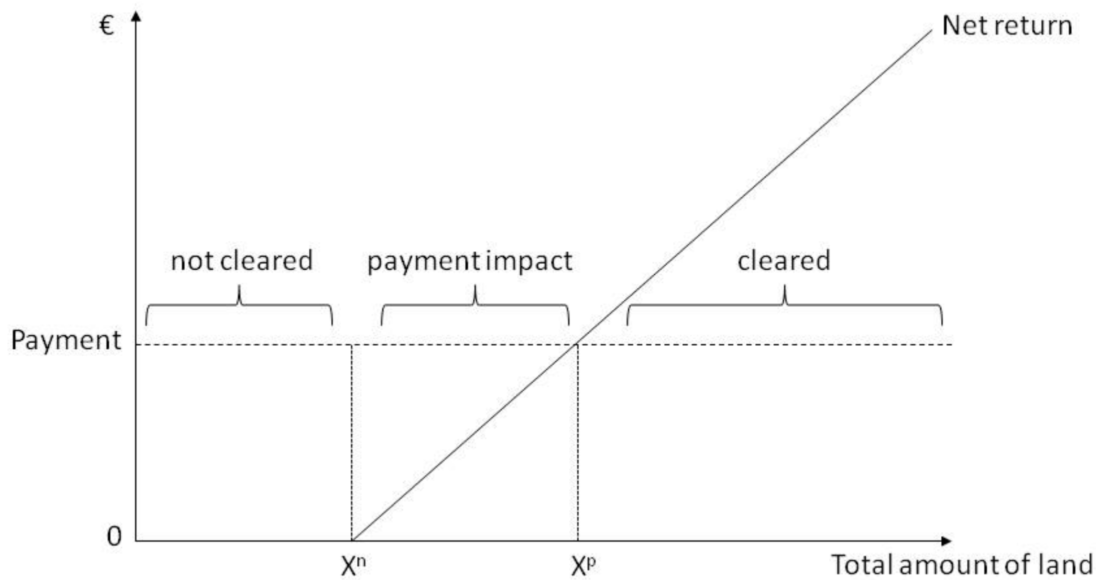
In order to be able to assess the impacts of an economic instrument, it is important to first of all have a good understanding of the instrument itself, its design, workings and embedding in the broader existing institutional-economic context (mix of policy instruments). A useful typology of policy instruments is found, for example, in Jordan et al. (2005). Economic instruments are also often referred to as market-based policy instruments, and typically include taxes, charges, levies, subsidies, and tradable permits. However, also voluntary agreements like many agri-environmental or agro-forestry agreements often include a payment mechanism (subsidy) laid down in contracts. The total effect of this policy mix on human behavior and associated land use changes will usually be the result of both the degree of voluntariness of the imposed policy instrument, the use of contracts to take away any uncertainties on the side of the land owner or land user about the payments they will receive in the future for the biodiversity and/or ecosystem services they provide on their land, and the level of payment. The next Section 4 provides a discussion of instrument selection and relevant aspects related to this selection.

One of the most important next steps are Steps 2 and 3 to identify the baseline conditions, i.e. the social and institutional-economic background situation and corresponding outcome(s) in which a financial-economic incentive is introduced. This crucial step will allow us to estimate the incremental change induced by the incentive in Step 4 compared to an existing or expected future situation and is referred to as the 'additionality criterion' in the PES literature. In cost-benefit analysis, this is usually referred to as the 'with' and 'without' situation, and in experimental economics as the control and treatment group.[2] A control group is used as a baseline measure. The control group is identical to all other items or subjects that you are examining with the exception that it does not receive the treatment or the experimental manipulation that the treatment group receives. The treatment group is the item or subject that is manipulated. For example, when examining the effect of PES on ecosystem service provision in a specific area, the researcher has two options to identify and quantify the effect of PES on provision and welfare. Either the same area is examined before and after the introduction of PES (where assumptions are made regarding the area's development without PES) or the area is compared to a more or less identical other area where no PES scheme is introduced (based on a more or less formalized matching procedure). In both cases, relevant boundary conditions have to be kept constant in order to filter out the incremental effect of PES on service provision and economic welfare (i.e. control for confounding factors that may also affect the outcome)[3]. The principle of additionality is illustrated with the help of Figure 2. An example of the impact of PES on deforestation rates in Costa Rica is provided in Box 2.

[2] Obviously, in an 'ex ante' analysis the comparison has to be based on expectations about future developments, whereas in an 'ex post' evaluation it can be based on observations.

[3] The latter is related to what is referred to as the 'conditionality criterion' in the PES literature, i.e. the condition that the agreed ecosystem service quantity and quality will be provided in the transaction.

Figure 2: Land use choices without and with a financial payment incentive

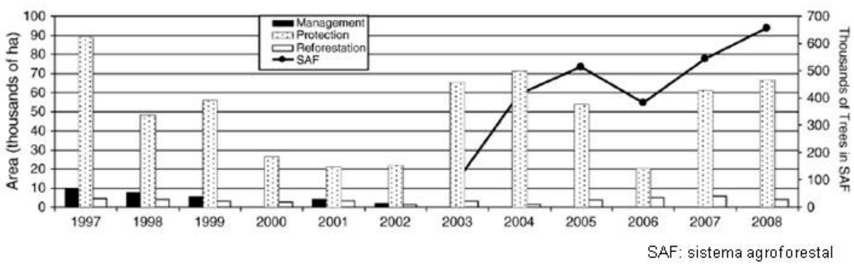


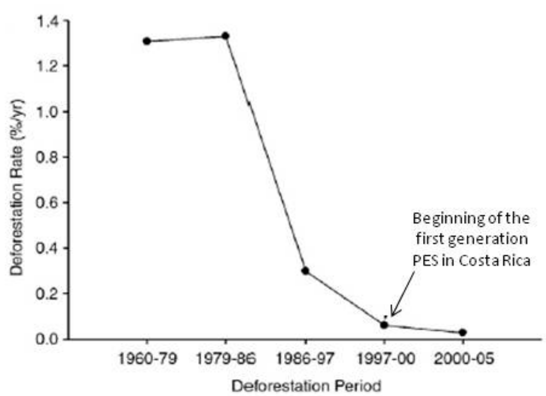
Source: Pfaff et al. (2008).

Box 2: Example of the importance of defining a baseline level in the evaluation of the impact of PES

In a critical review of the impacts of PES in ecosystem service provision, Daniels et al. (2010) find that PES has not lowered deforestation rates at the national level in Costa Rica. PES has a long history in Costa Rica, going back to 1979 when income tax credit was given to land owners involved in reforestation activities to offset the costs plantations. Since then, a variety of payment schemes have been implemented, including soft credits and financial compensation through tax vouchers. PES was authorized in Costa Rica since 1996 in the 4th national forestry law which recognizes 4 forest ecosystem services: biodiversity, watershed, scenic beauty and GHG mitigation through carbon storage and sequestration. Land holders participate through different land use modalities including reforestation through plantations, protection of existing forest, natural forest regeneration, and agroforestry systems. Payment per hectare is uniform across all contracts within each modality. Forest cover serves as a proxy for ecosystem services. When measuring the impact of PES, spatial data considerations, sampling considerations and the effects of institutional path dependency owed to the unique evolution of PES is essential. In one study, using a statistical matching approach to pair PES farms with non-PES farms that are similar in biophysical setting and market proximity, the 'matched' non-PES farms serve as a control representing the deforestation rate expected in the absence of PES.

The main outcome of this ('ex post') study is that payments for conservation had virtually no impact. Controlling for some of the other confounding drivers of deforestation, the analysis concluded that PES prevented forest loss on less than 0.25% of land enrolled in the PES program using matched non-PES farm deforestation as the main indicator. This means that forest would have been preserved on virtually all PES land even without payments. One important reason for this was that PES contracts were located in areas with very low probability for deforestation. By the mid 1990s when PES began there was relatively little deforestation to prevent compared with historical trends.





Source: Daniels et al. (2010)

A more positive assessment of the role of PES in reforestation in Costa Rica can be found in Arriagada (2008).

Figure 2 visualizes land holder choices regarding the available amount of land where payments for the ecosystem services provided by the land compete against the gains from non-forest land use. The latter are the returns net of costs of clearing and cultivating the land. Land will be cleared if net returns are positive. In the absence of a financial incentive not to deforest, land holders will clear forest land from X^n onwards where the net returns or net benefits become positive. Forest land will never be cleared in the interval [0-X^n]. Land owners will only participate in a PES scheme in the interval [0-X^p] where the payment is larger than any of the other opportunities. Not all who wish to participate will change their behavior though. Land holders in the interval [0-X^n] would not change their current land use as the area will stay forested without or with a financial incentive. Land holders in the interval [X^n-X^p] would deforest without any financial payment for ecosystem services, but not if they would be offered a payment higher than their 'opportunity costs', i.e. the net returns foregone related to crop cultivation.

Based on the outcomes in the with-situation (experimental treatment) and without-situation (control treatment), the relevant welfare implications have to be identified. This step is conducted in close collaboration between economists and ecologists. The actual or expected impacts on biodiversity and ecosystem service provision levels provide the basis for the estimation and valuation of the relevant economic values involved. The difference between welfare levels for the with and without situation is then compared to the additional costs of the introduction of the incentive. If different policy instruments or instrument design variations are available, the costs of each one can be estimated and compared to their actual or expected contribution to reaching the environmental objective in a cost-effectiveness analysis. Guidelines for the estimation of the costs of policy instrument design and implementation (Step 5) are found in section 5. This is followed in section 6 by the estimation of the welfare impacts of biodiversity conservation and ecosystem service provision (Step 6).

The assessment of the costs and benefits of the economic instrument (Step 7) through time typically takes place in a cost-benefit analysis (CBA) framework, where discounting of future flows of costs and benefits facilitates the comparison and identify to what extent the benefits outweigh the costs of introducing the economic policy instrument. Identifying the key assumptions in the analysis and expliciting these in a sensitivity analysis is an important final step (Step 8) to test the robustness of the outcome of the CBA.

4 Instrument selection in a policy mix

Before turning to the assessment of the costs and benefits of economic policy instruments, this section briefly discusses some of the key issues related to economic policy instrument selection. Different categories of economic policy instruments provide different incentives to change behaviour. Direct regulatory instruments set standards that people or firms have to comply with at the risk of being fined or prosecuted. Market-based instruments provide financial incentives. Informative instruments provide indirect incentives to people and firms by increasing transparency on all costs and benefits associated with the production and consumption of certain goods and services.[4]

Whichever instrument is selected by a policy-maker, if the instrument is designed carefully, then standard economic theory predicts that each instrument is able to achieve a given policy target. Differences in economic policy instruments only relate to the characteristics of the process of incentives and behavioural change that leads to the set target. As these characteristics differ, some instruments may reach a target at lower costs than others. Aspects like incomplete information, uncertainty and dynamic effects may lead a policy-maker to prefer one category of instruments over another, purely for reasoning based on economic theory (see, for instance, Baumol and Oates, 1988 or Goulder and Parry, 2008).

Uncertainty, dynamics, and incomplete information are also examples of factors that may disturb the simple choice for one policy instrument. They complicate the policy-maker's decision and in most cases make it impossible to reach the `first-best outcome'—the state of the world in which all market failures relevant to the problem at hand are solved. One solution to enhance efficiency in such a `second-best world' is to combine more than one instrument in a policy mix (see Ring and Schröter-Schlaack, eds. (2011) for a further discussion on this topic, and see Bennear and Stavins (2007)). This is not the only rationale for using a mix of policy instruments. Other reasons, some of which are particularly relevant for biodiversity protection, are the following.

Multiple objectives: If the policy-maker has multiple objectives in a certain area, both of which relate to one or more of the same resources or goods, a single instrument may not suffice. A classic example relates to the combination of poverty alleviation and biodiversity protection policies. Where poverty alleviation generally aims for economic development using the available natural resources, this may

[4] For more information, see the WP2 report (Ring and Schröter-Schlaack, eds., 2011).

contradict the objectives of biodiversity protection. A trade-off exists and an optimal policy has to weigh both policy goals and select instruments accordingly.

Multiple market failures: If a certain activity causes more than one market failure, and it is too costly to correct this market failure at the source, then the correction of these market failures where they impact requires more than one policy instrument (or, alternatively the use of the same policy instruments for two or more different targets). An example is a logging firm that causes both biodiversity loss and local erosion. When it is too costly, or infeasible due to incomplete information, to handle both problems with a single instrument, then this problem can be solved by e.g. combining a logging-charge (to prevent biodiversity loss) with a zoning regulation on areas where logging is prohibited (to prevent the local erosion problem).

Spatial considerations: Given information on opportunity costs of land, there may be spatial configurations for biodiversity protection that can best be targeted by a policy mix. For instance, when most gains can be made in one area, but not all. Then a park can be established in that designated area, and additional conservation can be reached by a more efficient instrument (e.g. PES) in the surrounding area. This is a case where there is a clear trade-off between transaction costs and costs of conservation.

Exogenous variability: In some cases, environmental damage depends on exogenous variability of a relevant factor. One example is air pollution where health impacts depend on wind characteristics and atmospheric conditions. Another example is water scarcity where damage depends on climatic conditions and may vary from year to year. In such cases, it is optimal to use a combination of instruments in order to efficiently handle the different types of problems that occur, depending on the exogenous parameters. It is clear that these two examples are only marginally related to the issue of biodiversity protection but there may be other, more relevant, processes where this exogenous variability plays a role.

In addition to the standard criteria used in instrument selection, recent research findings point to a wide array of other factors that should be taken into account. These factors are not necessarily in line with conventional thinking in economics on incentives and behavioural change. Three examples of such factors are motivation, trust, and legitimacy.

Motivation: People differ in their intrinsic motivation to perform a certain type of behaviour, say behaviour that is aimed at conservation of biodiversity. They may be intrinsically motivated because their behaviour corresponds to their environmental ethic or because they perceive biodiversity conservation as important. When a policy instrument is introduced that `rewards' such conservation behaviour, then this type of external motivation may (partly) replace the internal motivation. As a result, biodiversity conservation may not increase as much as expected or may even tend to decrease. This crowding-out effect has been observed for many types of `pro-environmental' behaviour and may differ depending on the instrument used (Frey and Stutzer, 2006, and Sommerville et al, 2010).

Trust: When local communities become involved in the protection of their natural resources (for instance through a PES or CBNRM programme), they are often assigned responsibilities for management, monitoring and/or enforcement. An increasing body of research suggests that such communities are able to overcome open-access problems themselves. The extent to which they succeed depends on a variety of characteristics, most of which point to the level of `social capital', of which trust is an important element (see Ostrom's research programme and e.g. Bouma et al, 2008).

Legitimacy: The extent to which local communities will comply with biodiversity policy instruments in place on their land depends to a large extent on the perceived legitimacy of these policies. Legitimacy is a broad term that captures many elements that relate characteristics of the policy to people's attitudes. For example, take a policy instrument that implements a zoning regulation in which a core zone is monitored by park rangers for people that collect NTFPs or go hunting in that zone. If the locals perceive the park's goals as legitimate, perceive the core zone as contributing to this goal, and respect the task of the park rangers, they will have an incentive not to violate the core zone restrictions, and vice versa. Hence the legitimacy of an instrument impacts its effectiveness (see e.g. Stern, 2008, Bouma and Ansink, 2013, Bouma et al., 2013).

In all PolicyMix case studies, the set of existing instruments that was already in place consists of a mix of direct regulation, usually in the form of protected areas, and economic incentives, usually in the form of PES or agri-environmental schemes. New instruments considered and/or selected for analysis are in most cases instruments that use economic incentives to persuade landowners to voluntarily implement conservation on their land. Specifics of each case study follow below.

In the Norwegian case study, a range of existing instruments (direct regulation, sector instruments, voluntary conservation, and incentivized conservation) has been assessed joint with proposed and potential new instruments (subsidy reform, fiscal transfers, auctions, and offsets). Interaction of instruments was assessed based on spatial overlap of existing instruments, and proposed instruments were preliminary assessed both mainly with regards to relations between and incentives of agents and stakeholders.

In the German case study, the main instrument being used is the implementation and management of protected areas using a variety of instruments at different scales (e.g. federal, länder, municipalities, with only minor funds available for a range of incentive-based instruments. New instruments considered include PES arrangements and ecological fiscal transfers (EFTs). Interaction of instruments may be hampered by a range of institutional constraints.

In the Finnish case study, a combination of regulatory instruments and privately protected areas based on a PES program co-exist. The area under the PES program (METSO) is small relative to the regulated area. A large range of upcoming and potential instruments was assessed, mostly based on incentives and alternative modes of land-use planning. Most of these are considered to be constrained by budgets, political will, and institutional rigidity. A survey on the uptake of the METSO PES program confirms this assessment.

In the Portuguese case study, direct regulation prevails combined with sectoral economic instruments that potentially affect conservation such as instruments used in land use and water management policy. In addition, Portugal has recently implemented a system of EFTs in addition to an agri-environmental scheme. Interactions between these instruments were assessed qualitatively.

In the Costa Rican case study, a national-scale PES program is dominant, with a strong focus on forest and silvopastoral systems. The program has been very popular with landowners. Based on experience with takeup and effectiveness, new approaches and financial mechanisms are considered within the program. Next to the PES program, Costa Rica is divided in 11 conservation areas, which includes national parks and other types of protected areas, all based on direct regulation. Finally, certification schemes and sectoral instruments affect conservation. A qualitative assessment of instrument interaction was performed which revealed both positive and negative interactions.

In the Brazilian case studies, direct regulation is mainly based on the national forest code, which stipulates minimum forest reserve areas. In addition, regional policies on protected areas and land use planning are in place. There are several examples of sectoral policies that contradict conservation targets, including some perverse agricultural subsidies and infrastructure projects. Finally, there is a range of economic instruments in place, including a rapidly increasing set of PES programmes. New instruments focus on expansions of PES programs and implementation of tradable development rights. Interactions between existing and new instruments are classified in a qualitative way. Interestingly, only few interactions are considered to be conflicting, while most are neutral or reinforcing/synergetic.

5 Cost assessment

5.1 Introduction

This section provides a concise set of guidelines to estimate the costs of using economic instruments as part of a policy mix for biodiversity protection and enhancement. Knowledge of these costs is essential in order to perform a cost-benefit analysis or a cost-effectiveness analysis with a view to selecting an appropriate instrument (mix). In particular, the claim that using economic instruments is an efficient biodiversity policy approach can only be substantiated if all relevant costs are taken into account.

A distinction can be made between:

Production costs, i.e. the opportunity costs of land use (refraining from certain land use practices that are profitable but harmful to nature) and the costs of the measures that have to be taken to protect biodiversity (e.g. creating favorable habitat conditions); and

Transaction costs, i.e. the costs associated with the introduction and application of the policy instrument(s) (e.g. creating legislation and institutions; gathering and exchange of information; entering into agreements and market transactions; monitoring and enforcement).[5]

Obviously, it is the sum of production and transaction costs that matters when deciding on policy instruments. Wätzold *et al.* (2010) use a slightly different cost concept framework, in which transaction costs are subdivided into implementation and decision-making costs (see Box 3).

Furthermore, the distinction between **ex ante** and **ex post** (and 'in medias res') analysis is also relevant when estimating the cost of biodiversity policy instrument use. The guidelines below are focusing on ex ante analysis. In general, this analysis will be the most challenging type in terms of data availability and uncertainties. However, also the estimation of costs afterwards (ex post) may be fraught by difficulties depending on the degree to which implementation and administration costs have been carefully monitored.

The approach presented can be used for single instruments as well as for a policy mix. The costs of a policy mix are equal to the sum of the instruments in the mix, minus the savings that can be obtained through synergies, plus any additional costs due to counteractive effects. One of the most important challenges is to find 'reasonable' (substantiated) proof of the incremental cost savings due to such synergies. Here too, the definition of the appropriate baseline conditions (without the policy mix) plays a major role in the identification of these synergy effects.

[5] One of the consequences of the existence of transaction costs is that market transactions which are potentially profitable for all parties involved may not materialize. This may for instance reduce the effectiveness of economic incentives (such as taxes and subsidies). Transaction costs can be a reason for applying a policy mix rather than a single instrument in environmental policy (see Lehmann, 2010).

Box 3. Costs of conservation

Wätzold *et al.* (2010) conducted a literature review combined with interviews of managers to scope the production, implementation and decision-making costs of the Natura 2000 protected area network in the EU (Box 3). Using a qualitative approach they focused especially on trade-offs between the different types of costs in several case studies. This scoping exercise can be recommended as a starting point for quantitative and GIS based opportunity cost estimates.

Wätzold et al. 2010 sub-divides total costs of conservation of the Natura 2000 network into production costs, implementation costs, and decision-making costs (p.2055):

Production costs are the costs of the actual conservation measures that are carried out including foregone economic benefits due to restriction on economic activities. Examples of production costs are costs for setting up and maintaining fences to protect reserves and foregone profits of farmers due to restrictions on farming for reasons of conservation.

Implementation costs include the costs of monitoring compliance with the law and—if necessary—of enforcement measures. Examples of compliance monitoring costs are costs for supervisory personnel and specialist equipment, while examples of enforcement costs are administrative costs for lawsuits and for collecting fines.

Decision-making costs arise from acquiring the information necessary for the successful design and implementation of conservation measures. This includes scientific and local knowledge about the effects of conservation measures on species as well as information needed for the cost-effective design of measures.

Opportunity costs

Protecting biodiversity can be costly. One of the main reasons is the fact that the conservation and enhancement of nature often precludes the use of the protected area for various profitable economic activities or requires restrictions on such activities. The net benefits foregone due to these prohibitions and restrictions are the 'opportunity costs' of the biodiversity project or policy. Assessing opportunity costs will mainly be relevant if the biodiversity policy consists of measures and instruments that reduce the opportunities for land use and development. Opportunity costs and their mapping are discussed in greater detail in separate 'Guidelines for opportunity cost evaluation of conservation policy instruments (POLICYMIX Technical Brief No. 11).

5.2 Steps in a cost assessment

Step 1: Identification

The analysis starts with a specification of the actions/activities, investments etc., that are necessarily (or likely to be) related to the use of the envisaged policy (instrument). Basically, this step consists of giving specific and detailed answers to the question: "What has to be done to make the instrument work as intended?"

The following broad categories of cost types can be helpful as a checklist:

- information costs;

- costs of planning, design and decision making;
- administrative costs;
- costs of monitoring and enforcement;
- indirect costs (e.g. delays, judicial procedures, market transactions).

Step 2: Selection

Assessing costs is costly in itself. Therefore, it is important to focus on costs that are likely to be significant. A preliminary screening of the cost components identified in step 1 can be done by estimating the order of magnitude of cost of each identified action. As a rule of thumb, the items with expected costs having an order of magnitude below the highest one can be ignored in the next steps, unless there are many of them (5 or more), or if the margins of uncertainty extend beyond the order of magnitude. The dropped cost items should still be mentioned qualitatively when reporting.

Step 3: Characterization

The main elements of this step are:

- identifying the actors (stakeholders, target groups, population) involved or affected (distinguishing between expenditure and actual cost bearing, taking into account transfers/subsidies);
- determining if the cost has a one-off or recurrent character (including timing / frequency);
- the nature of the cost: is it monetary or in-kind (e.g. man-hours)?

Step 4: Quantification

Once all the relevant and significant cost items have been identified and characterized, they can be quantified. How this is done depends to a large extent on data availability and reliability (see also the next section). In some cases, certain quantitative cost data may be directly measured (e.g. in an ex-post analysis or in an ex-ante analysis where conditions are very similar to an existing or previously applied policy). In other cases, the costs will have to be estimated, calculated or extrapolated using available data and 'reasonable' assumptions. Validation and cross-checking of data sources are important tools to improve reliability and to determine the margins of uncertainty (see also step 7).

Cost figures often have an aggregated or mixed character, i.e. only part of the cost is related to the specific policy under consideration. An example is the wages of officials involved in policy preparation or enforcement. Estimates of the part of their time devoted to the policy at hand will be needed to arrive at the correct attribution.

Step 5: Valuation (of non-monetary costs)

Just like benefits, information on costs will not always be directly available in monetary terms. An example is the time that farmers and foresters have to spend on filling forms and checking birds' nests. Usually, valuing such 'opportunity costs' will be less challenging than in the case of (environmental) benefits. For the working time spent by self-employed people, for instance, shadow prices can be applied by using available wage rate figures for equivalent qualified employees. For the working time of employees, a mark-up of 25% on gross salaries can be used to reflect overhead costs.

Step 6: Aggregation

For a complete CBA or CEA, costs will have to be aggregated over time. This requires the use of a time horizon and a discount rate. Obviously, in a CBA these parameters should in principle be the same for costs and benefits (unless there is a specific reason to apply a different discount rate to some items, such as long term irreversible impacts).

Step 7: Assessment of uncertainty and sensitivity analysis

Given that many cost items will be estimates rather than 'hard' data, the analysis will somehow need to deal with the uncertainty in the estimates. A sensitivity analysis can show the changes in outcome if the 'extreme' values of the range of possible cost figures are used (e.g. the values between which the real value is expected to lie with 90% likelihood).

The reliability of the results may be further enhanced by asking one or more independent experts to review them.

Finding useful data for the cost analysis can be a real challenge. Basically, there are two ways of obtaining them:

- Using available statistics, financial reports and other publications (e.g. estimates in impact assessments). This is usually the least expensive option (although finding the right sources may involve quite some searching time). The main disadvantage of this approach is that it is often hard to verify if the figures found are really relevant and appropriate for the analysis at hand. Often it will be unclear what exactly is covered by the figures, or they may not be representative for the type of policy under analysis. Moreover, the desired figures may not be available (e.g. because they are confidential or not collected at all) or not available at the level of aggregation that is needed.
- Using surveys, questionnaires, interviews etc. This option enables the analyst to ask directly for the information needed, and to approach the experts and stakeholders who are likely to possess this information. The obvious disadvantage of this option is that it will often be costly. Furthermore, one should be aware of the risk of bias in the responses (selective non-response, strategic behaviour etc.).

5.3 Instrument design and costs

Designing a biodiversity policy is likely to be an iterative process, in which information on costs and benefits of an instrument mix is used to improve its benefit/cost ratio in a next stage. Experience shows that the design of policy instruments can have a big impact on the costs of the policy. Important cost determining factors that should be taken into consideration when designing an instrument mix include:

- Scale. Many cost components are fixed, so large scale application will reduce average cost (e.g. per hectare). To the extent that costs are related to the number of actors involved, a low-cost design may require the exclusion of smallholders or the application of standard rules and conditions to them.

- Experience and learning. The cost of a particular instrument (mix) will usually decrease over time, as teething troubles disappear, experience grows and routines develop. Even though 'traditional' instruments that closely relate to existing practices and institutions will have a short term cost advantage, one should be aware of potential long term savings from 'innovative' instruments that may have relatively high initial costs.

- Existing legislative and institutional framework. Instruments that fit well into this framework may have a significant cost advantage. For example, incorporating biodiversity-friendly elements into an existing tax or subsidy scheme will entail huge savings compared to setting up a new, targeted scheme.

- Cultural and social factors. In social constellations with a tradition of cooperation and a high level of mutual confidence between stakeholders the cost of applying a certain instrument mix will differ from situations where conflict and confrontation are dominant features. In the latter case, the cost of the policy mix may become high due to e.g. obstruction, legal procedures[6] and the need for more intensive monitoring and enforcement. On the other hand, in more 'consensus-oriented' conditions, the cost of delays involved in introducing a new instrument may be higher due to protracted consultation and participation procedures. Generally speaking, there will often be a 'trade-off' between economizing on the cost of an instrument mix and its effectiveness in terms of targeted biodiversity protection.

- When estimating the costs associated with an instrument mix rather than a single instrument, one should of course be aware of the interdependencies and interactions between the instruments. Two different instruments may require similar administrative or monitoring activities, and these should obviously be counted only once if the two are combined in a policy mix. On the other hand, the use of one instrument may also increase the cost of using another one: for instance, a ban on hunting may increase the cost of obtaining information on biodiversity in the area that would otherwise be available as a free 'by-product' of the hunting activity.

[6] The risk of costly legal procedures may be reduced by using simple, unambiguous and clearly formulated rules and conditions.

- One should also be aware of the risk of double counting in cost assessments. An example is the charging of a fee for a license. To the extent that this fee covers the administrative cost of the authorities, it should only be counted as a cost to the licensee.

5.4 Experience with estimating transaction costs in the POLICYMIX case studies

In all PolicyMix case studies, the WP4 guidelines have been used to a larger or smaller extent. In two case studies, Costa Rica and Norway, actual estimates of transaction costs have been made. In most other case studies, WP4 guidelines have been used to assess the extent or relative size of opportunity costs and transaction costs of various policy instruments, given the context of the existing `policyscape', and given the institutional setting.

In the Costa Rica case study (Rugtveit et al., forthcoming) total transaction costs were estimated at 19.3% of the PES payments. The main part of this (18%) consisted of the costs of administrative tasks (such as monitoring and reporting), which are usually carried out by intermediaries ('regente forestal'). These intermediaries can charge a fixed percentage of 18% for these tasks.

In the Norway case study (Lindhjem et al., forthcoming), transaction costs were estimated as a fixed percentage of the opportunity costs: 20% for the voluntary program and 35% for the command-and-control program. On top of this, they accounted for the 'marginal costs of raising public funds', i.e. the costs of necessary taxation to collect funds for conservation. These were set at 20% of the sum of opportunity costs and transaction costs.

5.5 Relevant links and further sources of information

The 'Standard Cost Model' (SCM) is nowadays widely applied to assess the administrative costs of regulation. The application of the SCM is obligatory in Impact Assessments of EU legislation whenever a measure is likely to impose significant administrative costs. To this end, the EU's Impact Assessment Guidelines include an Annex (10) in which the SCM is presented. A more comprehensive SCM Manual can be found on the website of national SCM practitioners (www.administrative-burdens.com).

McCann *et al.* (2005, section 4) mention a number of example studies where the transaction costs of environmental policy instruments have actually been measured or estimated. Mettepenningen *et al.* (2009, 2011) provide estimates of (private and public) transaction costs of agri-environment schemes in Europe, based on surveys and direct measurement.

6 Economic valuation of biodiversity and ecosystem services

6.1 Introduction

In this section, first the concept of total economic value is explained in Section 6.2, including its components, followed by economic valuation approaches in Sections 6.3 and 6.4, distinguishing between market and non-market based methods. The various steps undertaken in an economic valuation study are outlined, and forest ecosystem services are linked to their different value types. Crucial to any economic valuation study is the biophysical underpinning of the economic values. The need for such an integrated approach is illustrated in the same table linking suitable biophysical and economic valuation methods to different forest ecosystem services. The scale at which an economic valuation study is carried out is paramount too, hence the reason why this addressed explicitly in Section 6.5. Meta-analysis is a powerful tool to synthesize outcomes and results in economic valuation studies. An overview and critical review of existing quantitative overviews of studies analyzing the economic values attached to forest ecosystem services is presented in Section 6.6. This is followed by a separate section (Section 6.7) on one of the most popular non-market valuation approaches in the literature since the past two decades, i.e. the application of choice experiments to assess the economic value of ecosystem services. Choice experiments have the convenient property that they also allow for the assessment of different economic instrument designs to test participation constraints of landowners or land users based on these different institutional-economic designs.

6.2 The concept of total economic value and its components

Economic valuation of (limited available) resources, be it human made or natural resources, has as a primary objective to inform policy makers about relative resource scarcity and guide economically efficient decision-making. Valuation is also considered to play an important role in creating markets for the conservation of biodiversity and ecosystem services. Such market creation requires three main stages: demonstration of values, appropriation of values, and sharing of benefits from conservation (Kontoleon and Pascual, 2007).

Economic values reflect what people are willing to trade-off to either employ or conserve these resources. Given the absence of a functioning market mechanism for many natural resources and ecosystem services, and in line with increasing conservation conflicts and a need for more efficient resource allocation, it is necessary to have knowledge and information of the marginal value or benefits of the resource in its alternative uses. An economic value is defined in terms of economic behaviour in the context of supply and demand. Put simply, it is the maximum amount of goods or services - or money income - that an individual is willing to forego (willingness to pay or WTP) in order to obtain some outcome that increases his or her welfare. The economic concept of value differs in this sense from the concept of value in other social sciences. It does not suffice to state that something is considered important, beautiful, worth protecting etc. Value is measured through people's (or society's) financial commitment to preserve something ('put your money where your mouth is'). Ideally, this commitment is measured through observed behavior in markets where people actually pay for a good or service. This actual behavior is considered to reveal people's preferences best and actual payments are considered the most reliable indicator of value.

If the outcome reduces welfare then this welfare loss is measured by the minimum amount of money that the individual would require in compensation (willingness to accept or WTA) in order to suffer the changes (see Box 5). These sums of money are demonstrated or implied by the choices people make, and thus reflect individuals' preferences for a particular change (e.g. employment or conservation of resources). It should be noted that the WTP measure of the impact on social welfare does not consider inequalities in the distribution of gains and losses among individuals. However, WTP is theoretically constrained by individuals' ability to pay. Aggregated across those who benefit from a good or service and hence will be affected by any change in their provision level, the aggregate WTP or WTA amount provides an indicator of their Total Economic Value (TEV). Environmental economists have introduced a taxonomy of this TEV, which captures the variety of values emanating from the different uses of resources (Figure 5). The main distinction is made between use and non-use values.

Box 5: Economic welfare measures

A distinction can be made between two types of welfare measures based on two different points of reference: the 'compensating surplus' (CS) and the 'equivalent surplus' (ES). The former equals the money income adjustment necessary to keep an individual at his initial welfare level <u>before</u> the change in the provision level of a good, while the latter equals the money income adjustment necessary to maintain an individual at his new welfare level <u>after</u> the change in the provision level of a good. Four relevant welfare measures associated with welfare gains and losses can thus be distinguished:

- WTP to secure a welfare gain (CS_{WTP})
- WTA to forego a welfare gain (ES_{WTA})
- WTP to prevent a welfare loss (ES_{WTP})
- WTA to tolerate a welfare loss (CS_{WTA})

The WTP measures have become the most frequently applied in valuation studies and have been given peer review endorsement, especially because they are constrained by income whereas WTA is not.

Use values

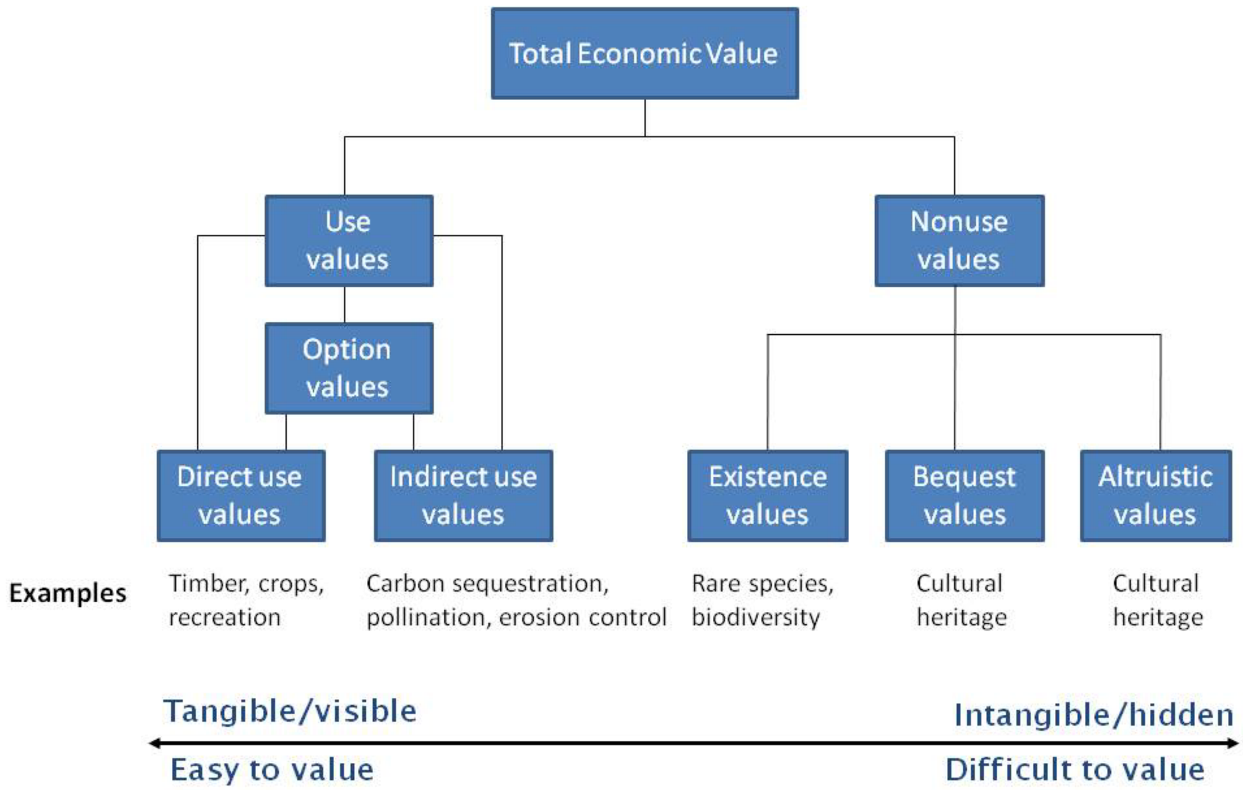
- Direct use values arise from direct interaction with natural resources. They may be consumptive, such as the extraction of timber from a wood, or they may be non-consumptive such as recreational activities or the aesthetic value of enjoying a view.

- Indirect use values are associated with services that are provided by natural resources but that do not entail direct interaction. They are derived, for example, from the prevention of soil erosion or pollination.

There is a further type of value that is related to *future* direct and indirect uses. This is option value:

- Option value is the satisfaction that an individual derives from ensuring that a natural resource and its services are available for the future given that the future availability of the resource is uncertain. It can be regarded as an insurance for possible future demand for the resource.

Figure 4: Components of Total Economic Value



Nonuse values

Non-use value reflects value in addition to that which arises from usage and is derived from the knowledge that a natural resource is maintained. By definition, nonuse values are not associated with tangible benefits that can be derived from it (though resource users may derive non-use values). Thus individuals may have little or no use for a given natural asset, but would nevertheless feel a 'loss' if it would disappear. Non-use values are linked to ethical concerns and altruistic preferences. However the boundaries of the non-use category are not clear cut and some human motivations which may underlie the position that the asset should be conserved 'in its own right', and labeled existence value, are arguably outside the scope of conventional economic thought. In practice, what is at issue here is whether it is meaningful to say that individuals can assign a quantified value to the environmental asset, reflecting what they consider to be intrinsic value.

Non-use values can be divided into three types of value (which can be overlapping): existence value, bequest value and altruistic value.

- Existence value is the satisfaction derived from a resource continuing to exist, regardless of whether or not it might be of benefit to others. Motivations here could vary and might include having a feeling of concern for the asset itself (e.g. a threatened species) or a "stewardship" motive whereby the "valuer" feels some responsibility for the asset.

- Bequest value is the satisfaction derived from ensuring that a resource will be passed on to future generations so that they will have the opportunity to enjoy it in the future.

- Altruistic value is the satisfaction derived from ensuring that a resource is available to contemporaries in the current generation.

It is important to note that what is being valued is not the ecosystem per se, but rather independent elements of goods and services provided by biodiversity and forest ecosystems. The aggregation of all function based values provided by a given ecosystem yields the TEV of that ecosystem. TEV does furthermore not provide an exhaustive assessment of the value of natural resources and ecosystem services to society. It measures the extent to which goods and services provided by ecosystems touch on the welfare of society, as direct determinants of individuals' wellbeing or via production processes. It represents two fundamental sets of values: individual values and production values. Individual values include recreational and amenity values, as well as non-use values (existence, bequest and philanthropic values) of goods and services provided by ecosystems. Production or output values occur through the employment of natural resources and ecosystem services as 'natural capital' in the production of other goods and services.

Forest woodlands are natural assets that create flows of goods and services over time. The key to their valuation is to establish the functions that they provide and link this to states or outcomes that are valued by society. If that link can be established, then the concept of derived demand can be applied. The value of a change in the functions provided can be derived from the change in the value of the stream of benefits. Given the multi-faceted nature of benefits associated with forest ecosystems there is a need for a useable typology of the associated values.[7] The focus here is on economic values, which depend on human preferences, i.e. what people perceive as the impact on their welfare. Economic values are relative in the sense that they are expressed in terms of something else that is given up (the opportunity cost).

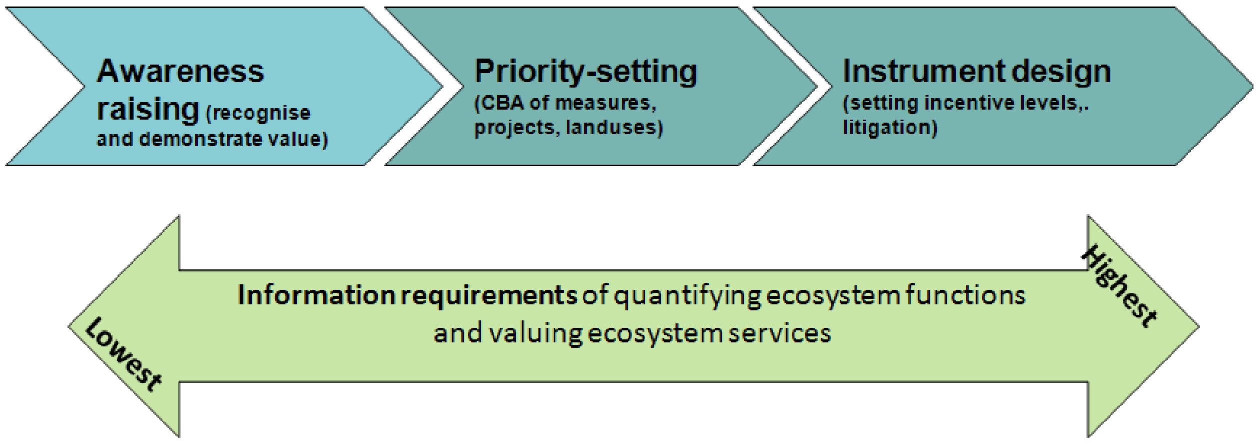
Figure 5 illustrates three broad types of policy contexts where economic valuation estimates are used. When valuation estimates are used for the purpose of awareness raising – recognising and demonstrating value – requirements for accuracy and reliability are at their lowest, as are information costs. A broad requirement is that valuation estimates show that values of ecosystem services are significant and different from zero.

For priority-setting in cost-benefit analyses of measures, projects or landuse needs for accuracy and reliability are greater. Valuation estimates need to answer questions of whether benefits are significantly greater than costs, and whether net benefits of one alternative are significantly greater than another.

[7] See the WP3 guidelines (Rusch *et al.* (2011), p. 24) for a typology of ecosystem services (processes and benefits) and its relationship with the classification followed by the Millennium Ecosystem Assessment (into supporting, provisioning, regulating and cultural services).

For instrument design, including setting incentive levels in PES, or in determining environmental litigation amounts in legal proceedings, the need for accuracy is at its highest as are the information costs. Valuation estimates have to be accurate in their 'absolute' levels.

Figure 5 Policy motivations for valuation and information needs



6.3 Valuing ecosystem services and biodiversity

The valuation of ecosystem services is a subject of growing enthusiasm and academic debate as the number of valuations studies and meta-analyses continues to grow. In 1997 Costanza *et al*. attempted to value ecosystem services using various case studies to derive average values per hectare for 17 ecosystem services from 16 different biomes. These values were then extrapolated by multiplying these values by hectares of each biome to obtain global values. Costanza *et al*. estimated the aggregate value of global ecosystem services to range from $ 18-61 trillion, averaging $ 38 trillion (updated to 2000 USD) and noted that a large part of this value was due to non-marketed ecosystem services. In this paper forest values were estimated at roughly 4.7 trillion USD as the aggregate flow value. This figure was further broken down to values for both tropical and temperate forest types. Among the disaggregated values there were high climate regulation values for tropical compared to temperate forests, high values attributed to nutrient cycling in tropical forests, along with a large part of the value attributed to raw materials and recreational benefits. Temperate forests were estimated to have a slightly higher value for food production but produce a much lower aggregate value of 894 billion USD compared with 3.8 trillion USD for tropical forest value.

The overall value for global ecosystem services derived by Costanza *et al.* is similar in size to global Gross National Product and has been criticized by some in the economic community including Costanza himself who freely admitted faults in the study, including 1) assumed homogeneity in natural capital and economic contexts, 2) the study being partial and static rather than dynamic and 3) values taken from various studies varied greatly in methodology, practical and theoretical assumptions (Costanza et al, 1998).

According to Nunes and Van den Bergh (2001) valuation of biodiversity can be divided into genetic, species, ecosystem and functional diversity. Genetic diversity refers to the level of variation among genes present in a certain species while species diversity is defined by the number of species present. Ecosystem diversity refers to the number of communities present in the habitat, and landscape level as well as physical conditions present. Functional diversity can be defined as an ecosystem's ability to withstand change and shock without experiencing a tipping point or regime shift that results in a qualitatively different state of the ecosystem; generally referred to as ecosystem resilience and adaptive capacity (Huitric, 2009).

6.4 Economic valuation approaches

An important distinction to make is between market-based valuation techniques and non-market based valuation techniques. Market valuation means that existing market behaviour and market transactions are used as the basis of the valuation exercise. Economic values are derived from existing market prices for inputs (production values) or outputs (consumption values), through more or less complex econometric modeling of dose-response and/or damage functions. Examples include the economic value of timber, which is sold on a market (market analysis), the costs of soil fertilization to compensate for soil erosion (restoration costs) or water treatment due to soil runoff and sedimentation (damage cost), or the costs of a water filter on tap water (avertive behaviour/defensive expenditures).

The economic value of ecosystem services provided by forests and woodlands can be measured directly through existing market prices for intermediate or final products (e.g. timber price). Here, the market price is multiplied by the quantity of timber sold or consumed to yield the total market value. The market price may have to be adjusted to provide the real economic shadow price, but otherwise it is likely to provide a relatively simple means of assessing economic value. In some cases, human resource use may also include recreational activities. Examples are walking, cycling, camping, or undergoing a wildlife experience. In some cases recreational values can be derived from existing entrance fees.

Many forest ecosystem services are not traded in markets and therefore remain un-priced. It is then necessary to assess the economic value of any environmental damage (avoided with the help of existing pollution abatement and mitigation measures) with the help of *direct* and *indirect* non-market valuation methods. Non-market valuation means deriving economic values in cases where such markets are non-existent or distorted. Direct methods (also called stated preference methods) refer to contingent valuation (CV), discrete choice experiments (CE), and contingent ranking (CR) techniques, where individuals are asked directly, in a social survey format, for their WTP for a pre-specified environmental change. WTP can also be measured indirectly by assuming that this value is reflected in the costs incurred to travel to specific sites, such as with recreational visits (travel cost studies), or prices paid to live in specific neighbourhoods (hedonic pricing studies) (also called revealed preference methods). The latter two approaches are based on preferences being 'revealed' through observable behaviour, and are restricted in their application to where a functioning market exists. CV, CE and CR, being based on surveys that elicit 'stated preferences', have the potential to value benefits in all situations, including non-use or passive use benefits that are not associated with any observable behaviour. The legitimacy of these methods and results is still contested, especially in the context of non-use values, and conducting surveys can sometimes be a lengthy and resource-intensive exercise. Of these methods, CV is probably the most widely applied method in contemporary valuation research, but since about 10 years the use of choice experiments has increased exponentially too.

Table 1 provides an overview of the ecosystem services provided by forests and woodlands, the associated (direct and indirect) use, option and nonuse values and appropriate valuation approaches. The classification of ecosystem services (provisioning, regulating, cultural and supporting) is based on the Millennium Ecosystem Assessment (MA, 2005).[8] The steps in conducting an economic valuation study are presented in Box 6 below.

Box 6: Steps in the economic valuation of environmental goods and services
(Source: adapted from Brouwer, 2000)

1) Identification of the goods and services provided by the resources amenable to robust valuation

2) Assessment of their provision (target) level, including quality attributes, compared to the baseline (reference) level of provision

3) Identification of the groups of people in society (users and non-users) who benefit from the goods and services involved or who will be suffering a loss when they are removed, destroyed or degraded

4) Identification of the possible values (use and non-use values) attributed to the goods and services involved by these groups in society

5) Selection of the appropriate economic valuation technique(s)

6) Estimation of the economic value of the change in provision level of the goods and services involved, accounting for substitution and income effects and other contextual factors

7) Quantification of the 'market size', that is, the total population of beneficiaries over which the economic value is aggregated, accounting for possible distance-decay effects (people living further away may attach less value to the goods and services involved)

8) Estimation of the total economic value of the change in the provision of the environmental goods and services associated with the policy change

Table 1: Forest ecosystem services, value type and appropriate biophysical and economic (e)valuation methods

[8] A more detailed classification can be found in the WP3 guidelines (Rusch *et al.*, 2011), which also contains a figure showing the correspondence with the MA classification (Figure 5 on page 24).

Ecosystem service	Direct use value	Indirect use value	Option value	Nonuse value	Biophysical evaluation method(s)	Economic valuation method(s)
Provisioning services						
Food production	+	-	+	-	Yield (ton/ha/year) Crop yield model	Net present value of crop yield Market-price valuation
Timber production	+	-	+	-	Timber yield (cubic meters/ha/year) Forest yield model	Net present value of timber yield Market-price valuation
Water supply	+	+	+	-	Water extraction (cubic meters/year) Hydrological (water balance) model	Market value Market price valuation
Bioenergy production	+	+	+	-	Yield (ton/ha/year) Crop yield model	Market value Market price valuation
Regulating services						
Carbon sequestration	-	+	+	-	Carbon sequestration (tons carbon/ha/year) Carbon accounting model	Multiple approaches including damage cost avoided or EUTS values
Water quality regulation	-	+	+	-	Water quality model, biochemical water quality indicators (eg mg N, BOD or Chlorofyl/liter)	Avoided opportunity costs
Erosion control	-	+	+	-	Land use model? (tons of soil eroded per year)	Damage cost avoided, avoided opportunity costs
Pollination	-	+	+	-	Production function where bee density is an input factor in crop production	Production function method
Flood regulation	-	+	+	-	Probability of flood happening in location GIS-based hydraulic-hydrological model	Damage cost avoided
Cultural services						
Recreation	+	-	+	+	Landscape properties with known positive	Travel cost, hedonic pricing, contingent

Ecosystem service	Direct use value	Indirect use value	Option value	Nonuse value	Biophysical evaluation method(s)	Economic valuation method(s)
					or negative recreation potential	valuation, choice experiment
Tourism	+	-	+	+	Visitors/year, overnight stays	Tourist expenditures Market price valuation
Landscape aesthetics	+	-	+	+	Landscape properties with known positive or negative perceptual values	Travel cost, hedonic pricing, contingent valuation, choice experiment
Cultural heritage	+	-	+	+	Number and area of significant sites	Travel cost, hedonic pricing, contingent valuation, choice experiment
Supporting services						
Soil formation	-	+	+	-	Soil-yield production model (tons of soil/year)	Avoided opportunity costs
Nutrient cycling	-	+	+	-	Nutrient balance model (kg N/year)	Avoided opportunity costs

6.5 Scale issues in the valuation of ecosystem services

To address the provision of ecosystem services in an adequate way, it is important to consider the different spatial scales at which they are generated, supplied and valued by stakeholders. Ecosystem services affect stakeholders at different scales in a positive or negative way. For example, a national policy to transform timber forests to natural forests may cause conflicts between two scales (national: benefits of climate regulation vs. local: loss in income from timber sales) and within one scale (e.g. local enterprises: loss in income from timber sales vs. residents: recreational benefits). Many policies for biodiversity conservation create an imbalance between costs arising at the local scale and benefits provided to the national scale. Exploring the associated benefits and costs helps to anticipate and address potential conflicts between stakeholders at different scales. Information about scale issues is highly relevant for the appropriate design of new or assessment of existing policy instruments to protect biodiversity and ecosystem services. The economic analysis of instruments in a policymix (*ex ante* and *ex post*) thus requires that economic valuation methods take spatial scales into account. To date ecosystem service valuation appears limited in this respect. The majority of studies focus on one specific scale (e.g. a local forest) and merely assess the value of the service it provides at that same scale (Pascual and Muradian, 2010; Hein et al., 2006).

The challenge of economic analysis and valuation is to identify at which scales certain ecosystem services are relevant and to value the benefits and costs they generate for different stakeholder groups (Millenium Ecosystem Assessment, 2003; Turner et al., 2003). Table 2 illustrates the relevant biodiversity and institutional scales at which stakeholders may be affected. The grey boxes give examples of the scales involved, ecosystem services and values that would be relevant to an increase in natural forest area in a specific landscape.

Table 2 Scales of biodiversity, ecosystem services, and institutions (stakeholders)

Biodiversity scale	**Ecosystem service**	**Institutional/stakeholder scale**
Global	Climate regulation	International
Biome	Climate regulation Flood/erosion control	National economy Sector
Landscape/ecosystem	Climate regulation Flood/erosion control Timber provision	Municipality Community
Plot Single (group of) plants, animals etc.	Carbon sequestration Recreation Flood/erosion control	Household Firm

In order to integrate these scales in the valuation of ecosystem services and corresponding assessment of biodiversity policies, the classical framework of economic valuation (see Box 6) should pay attention to the identification of the spatial scales at which the ecosystem services under investigation are supplied to stakeholders.

This analysis of scales should include the following:

- ecosystem services generated at a certain biodiversity scale can be provided to a range of stakeholders at different institutional levels;
- stakeholders at a certain institutional scale can obtain ecosystem services generated at a range of biodiversity scales;
- within one institutional scale there might be different stakeholder groups (winners and losers).

Table 2 presents the most relevant biodiversity and institutional scales and can guide the analysis of biodiversity scales and stakeholders. A range of methods are available to include spatial scale issues in the economic analysis of ecosystem services and policy instruments.

In order to gain insight into potential conflicts of interest (winners and losers) within and between institutional scales a stakeholder analysis is a useful first step of the economic analysis (Pascual and Muradian, 2010). If conflicts among stakeholders exist, stakeholder-oriented valuation approaches (e.g. deliberative valuation) can help to identify and possibly solve such conflicts, as they are designed to value different policy options and eventually reach consensus about the most appropriate policy instrument (Pascual and Muradian, 2010, Spash, 2008). Stakeholder-oriented valuation can be used both *ex ante* for policy design and *ex post* for the assessment of a certain policy instrument. If the aim is to quantify stakeholder benefits in monetary terms, Choice Experiments can be designed in a way to account for differences in the spatial distribution of environmental services and beneficiaries (see Brouwer, et al., 2010). Respondents living in different parts of a country can be asked to value changes in forest quality/quantity in different parts of the country. This spatial addition to the choice sets also helps to recognise possible substitution effects. Spatial scales can also be introduced to Benefit Transfer, but this enhances the complexity of transferring values between sites considerably. In order to integrate spatial scales, Benefit Transfer is combined with geographical information systems (GIS). Such analyses include spatial variables, socio-economic characteristics of stakeholders (e.g. income and preferences) (Bateman, et al., 2003).

6.6 Meta-analysis of existing forest ecosytem services valuation studies

Over the past 8 years, 5 meta-analyses have been conducted, summarizing and explaining the results from existing forest valuation studies. Meta-analysis is a statistical method to detect systematic variation in observed outcomes for example of forest valuation studies, which produced the highest economic value flows. These studies are summarized in Table 3. Two only use stated preference studies such as CV and CE, one focuses on travel cost studies only, one on both CV and travel costs and one contains a mix of stated, revealed, market prices, cost estimates and even benefits transfer values. Bateman and Jones (2003) are the only study that distinguishes between different value types estimates in the different studies examined in their meta-analysis. Use and option values yield a significant higher forest recreation value than nonuse values. An overview of the different variables tested in each meta-analysis is presented in Table 4. The plus and minus signs in Table 4 refer to the direction of the marginal effect of each variable tested in the meta-analysis and the asterixes to the statistical significance level of the effect. A blank cell implies that the relevant variable was not tested in the particular meta-analysis study.

An important contribution of the valuation literature (and WP4) is to address feasibility of financing conservation.Valuation studies to a large degree tend to focus on the “financing instrument” side of conservation policy (WTP studies), whereas only a few studies look at the incentive effects of different types of compensation/payment mechanisms (WTA).

Table 3: Existing meta-analyses of forest valuation studies

	Bateman and Jones (2003)	**Lindhjem (2007)**	**Zanderson and Tol (2009)**	**Barrio and Loureiro (2010)**	**Ojea et al. (2010)**
Focus	Woodland recreation	Non-timber benefits	Forest recreation	Forest values	Forest values
Scale	UK	Scandinavia	9 EU countries	World	World
Number of studies	30	28	26	35	65
Valuation method(s)	CV, TC	CV, CE	TC	CV	Mix
No. of value estimates	44	72	166	101	172
Explanatory power (R^2)	0.643	0.815	0.851	0.896	0.617

All meta-analyses have a high explanatory power varying between 60 and 90 percent. Bateman and Jones (2003) and Zanderson and Tol (2009) both focus on forest (open access) recreation values. The three other meta-analyses focus on forest values more generally (Lindhjem (2007) on non-timber benefits). Barrio and Loureiro (2010) and Ojea et al. (2010) explicitly distinguish between different ecosystem services in their analysis. The former control for wooduse and recreation, while the latter distinguish between the Millenium Ecosystem Assessment (2005) ecosystem services: cultural services, provisioning services, regulating services and a mix of the three. Only recreation is statistically significant in Barrio and Loureiro and has a positive effect on stated WTP. Commercial wooduse does not have a significant effect. However, it is not clear what exactly the baseline category is in their study. In Ojea et al. regulating services are the baseline category and only those studies that focus on a mix of services appear to have a significant positive effect on forest values. No significant differences are detected between single category ecosystem services.

Species composition has been shown to have a positive impact on the recreational choice of forests by increasing the popularity in forests with a higher diversity of species compared to forests with lower diversity (e.g. Scarpa et al. 2000). Zanderson and Tol (2009) show that in recreational studies, visitors more generally prefer open forests with diverse tree age stands and smaller rather than larger sites. However, they are unable to detect a significant effect for the diversity of tree species, measured through the Shannon index[9]. Similar sensitivity to scope effects where marginal WTP decreases as the

[9] The Shannon index of diversity takes into account the richness and evenness of species distribution. The higher the index, the more rich and evenly distributed the species classes. Also age diversity can be measured with the Shannon index and this appears to have a significant impact on recreational forest values in Zanderson and Tol (2009).

size of a forest increases are found in Ojea et al. (2010). Lindhjem (2007) includes a dummy in his analysis if forest size is not mentioned in a valuation study. This appears to significantly inflate stated WTP.

Like Zanderson and Tol (2009), also Barrio and Loureiro and Ojea et al. include biodiversity indicators in their analysis. Barrio and Loureiro (2010) include dummy variables in the analysis if the valuation study focused on either flora or fauna and a separate dummy variable labeled 'biodiversity' if the study valued both flora and fauna. Ojea et al. (2010) also include dummy variables for flora and fauna separately, but use different background indicators for flora and fauna, one based on the IUCN Listed Species (absolute biodiversity indices of flora and fauna) and one based on the IUCN Red Species (relative biodiversity indices of flora and fauna). Only the listed species indicator for fauna yields a positive significant effect. The model that included the Red Species indicators only has a significant effect for flora, but this effect is negative. Because the listed species indicator yields a better statistical fit, this indicator is used in an extended model including interaction terms with ecosystem services. This reduced the significance level of the positive fauna indicator from 5% to 10% and produced a significant positive interaction between the flora indicator and provisioning services at the 10% level. None of the other interaction terms are significant. The interpretation of this positive interaction term could be that a higher plant species abundance level is related to higher forest economic values from provisioning services, including timber and non-timber forest products.

Table 4: Overview effects of variables tested in existing meta-analyses of forest valuation studies

Study characteristic	Variable	Bateman and Jones (2003)			Lindhjem (2007)			Zanderson and Tol (2009)			Barrio and Loureiro (2010)			Ojea et al. (2010)	
	Constant	+	***		+	**		+	***						
Location	Sweden				+	**									
	Finland				+	*									
	Scandinavia										ns				
	Europe										-	***			
	USA										ns				
	Other countries										ns				
	Latitude							ns							
	Urban				ns						ns				
	Open land							+	***						
Year	Study year				+	**		ns							
	Publication year													ns	
	<1995										ns				
	1996-2002										+	*			
Forest characteristics	Old growth										ns				
	Rainforest										+	***			
	Other forest types										+	***			
	Boreal													ns	
	Temperate coniferous													+	*
	Temperate mix not coniferous													ns	
	Tropical wet													ns	
	Tropical dry													ns	
	Forest size				ns			-	***		ns			-	***
	Size squared							-	*						
	Forest size in country										-	***			
	Share national productive land				ns										
	Share national land				ns										
	Local good				ns										

Study characteristic	Variable	Bateman and Jones (2003)			Lindhjem (2007)			Zanderson and Tol (2009)			Barrio and Loureiro (2010)			Ojea et al. (2010)	
	Regional good				+	*									
	Hotspot (protected area)													ns	
Ecosystem services	Wood use										ns				
	Recreation										+	***			
	Cultural													ns	
	Provisioning													ns	
	Mix cultural-provis.-regulating													+	***
Biodiversity	Flora										ns			ns	
	Fauna										ns			+	*
	Flora and fauna										ns				
	Species diversity							ns							
	Age diversity							+	***						
Management practices	More cautious				ns										
	Mix of practices				ns										
Value types	Use				ns										
	Use and option	+	***												
	Avoid loss				+	*					-	*			
	Gain change										ns				
	No change										ns				
Economic instruments	**Payment mode**														
	Voluntary donation				+	***									
	Recreational use payment				ns										
Population characteristics	GDP/capita							-	***		+	***			
	GDP													ns	
	Population density							+	***					+	**
Valuation method	Market prices													ns	
	Revealed preferences													ns	
	Other valuation													ns	
	Choice experiments				ns										

Study characteristic	Variable	Bateman and Jones (2003)		Lindhjem (2007)		Zanderson and Tol (2009)		Barrio and Loureiro (2010)		Ojea et al. (2010)	
	Actual payment					-	***				
Travel costs	Average distance					+	***				
	Cost/km					+	***				
	Opportunity cost time					+	**				
	Expenditures					ns					
	Holiday					ns					
Survey method	Personal interviews							ns			
	Mail survey			-	***[ʃ]			+	***		
	Other survey							ns			
	Sample size							ns			
CV elicitation format	Open ended	-	**	ns				+	**		
	Payment card	ns[ʃʃ]		ns							
	Dichotomous choice							+	***		
	Iterative bidding	ns									
Payment unit	Individual			+	***			-	***		
Payment frequency	One-time			ns[ʃʃʃ]				ns			
	Annual permanent							+	**		
	Annual temporary							ns			
Publication effects	Published paper			ns							
	Thesis			-	***	-	***				
	Author effects	+	***			+	***				

ns not significant; * significant at 10%; ** significant at 5%; *** significant at 1%

[ʃ] In combination with response rates and season survey.
[ʃʃ] Payment card with high range values appeared to inflate WTP at 10% level.
[ʃʃʃ] Payment mode was mixed up with payment frequency. The latter had no significant effect.

Only Barrio and Loureiro and Ojea et al. control for the type of forest in their analysis. Barrio and Loureiro find a significant positive effect of rainforest compared to coniferous forest on WTP values, but not for old growth. Ojea et al. on the other hand find a significant positive effect of temperate coniferous forests on values compared to Mediterranean forest. Tropical forest is only significant at the 10% level in their reduced model excluding interaction terms between the biodiversity indicators and ecosystem services. However, due to the use of different baseline categories, the results between the two meta-analyses are also here not directly comparable. If the forest is located in an urban area, this does not seem to have any effect as demonstrated in Barrio and Loureiro and Lindhjem, neither are protected areas (hotspots) significant in Ojea et al. (2010).

A remarkable and at the same time worrying finding is that despite the use of a wide variety of different valuation methods, measuring different types of use and nonuse values, Ojea et al. do not find a significant effect between valuation methods. Also Lindhjem does not find a significant effect between CV and CE. The application of different types of economic instruments, generally referred to as payment vehicles in the stated preference literature, has only been tested in one study (Lindhjem(2007)). If visitors are asked for a voluntary donation, this has a significant positive effect on stated WTP. This outcome differs from the findings in Brouwer et al. (1997) who show that an increase in annual income tax where everybody pays significantly increases WTP. Although in line with common practices in wildlife conservation, voluntary payments tend to increase protest rates for public environmental goods, partly due to concerns of free riding (Brouwer and Slangen, 1998). Contrary to Barrio and Loureiro (2010), Lindhjem finds no effect of the payment frequency on stated WTP (annual or one-time-off). The former show that if payments are annual and permanent, this too has a significant positive effect on stated WTP in CV studies. Hence, few meta-analyses of forest ecosystem services investigate the influence of payment vehicle on WTP while selection of an appropriate payment vehicle is imperative to create a realistic and acceptable method of securing payment as in Payments for Ecosystem Services (PES) schemes.

Finally, there are a number of results in one analysis, which are contradicted or show the opposite sign in another. This includes, for example, the impact of GDP per capita, which is expected to have a positive effect on travel behavior and stated WTP, but is only significant and positive in Barrio and Loureiro, not in Ojea et al. and negative in Zanderson and Tol. On the other hand, Barrio and Loureiro find a positive effect on stated WTP if an open-ended elicitation format was used, which goes against common findings in the CV literature that open-ended WTP values generally produce significantly lower values. If a respondents answers the WTP questions in a stated preference survey as an individual instead of on behalf of his entire household, this has - as expected - a significant negative impact on WTP in Barrio and Loureiro, but a positive effect in Lindhjem.

6.7 Choice experiments

Over the past 2 decades, choice experiments are increasingly used in the environmental economics domain to assess public perception and valuation of ecosystem services provision. Choice experiments have become one of the dominant valuation approaches in the environmental valuation literature in the past decade since the end of the 1990s. Choice experiments have a number of advantages compared to contingent valuation. These advantages refer primarily to a more detailed description of the goods and services involved and variations thereof. In a choice experiment respondents are typically asked a series of repeated choice questions aimed at evaluating different project or policy alternatives in terms of their key characteristics. All they are asked to do is indicate which alternative they prefer. Based on the choices respondents make, the researcher is able to derive the marginal utility associated with each attribute. If a monetary price is included as one of the attributes, this allows the researcher to express the marginal utilities into a monetary marginal WTP or WTA value. By including a price into a bundle of other good or service characteristics, the respondent is asked to better consider the trade-off between different attributes and attribute levels and the price involved. This adds to the cognitive burden of the stated preference exercise, but at the same time also controls for some of the difficulties and biases introduced by different WTP elicitation formats in continent valuation research. An important advantage of choice experiments is that it allows respondents to get acquainted and experienced with otherwise unfamiliar decisions (learning), which is expected to increase the validity and reliability (and accuracy) of the stated WTP values. Discussions of their challenges and applicability in different policy domains are provided in Bennett and Birol (2010) and Otieno (2011).

An important advantage of stated choice experiments is that it allows both monetary valuation of the nonmarket values of ecosystem services provision, and the institutional-economic context in which these services are provided (Bennett and Birol, 2010). Examples of the use of choice experiments specifically related to public preferences for PES are provided, for example, in Bienabe and Hearne (2006), Schaafsma et al. (2009), Kaczan et al. (2011) or Zander and Garnett (2011). Specific applications of choice experiments to inform contract design are provided, for example, in Brouwer and Akter (2010) in the context of micro flood insurance and Tesfaye and Brouwer (2011) in the context of sustainable soil conservation.

A comparison of choice experiments focusing on forest biodiversity and ecosystem service valuation (see Table 5), shows that a wide variety of characteristics is used to describe the policy or scenarios. Very few include specifications of the institutional-economic context in which the policy scenarios will be implemented. Only the two studies conducted in Finland by Lehtonen et al. (2003) and Horne (2006), discusses explicitly the contractual arrangements with the ecosystem service provides in the choice experiment. However, the payment mode differs in these 2 studies. The former is a WTP study, asking the public to pay through an increase in annual income taxation, whereas the latter concerns farmer/land owner WTA compensation, where the compensation is a constant value per hectare per year. In most studies eliciting public WTP for forest conservation, general income taxation is applied as the payment vehicle. A number of studies (4 out of the 15 in Table 5) do not further specify the payment mode. In 2 studies in Africa, a levy on land holdings are used and an entrance fee for tourists to access recreational parks.

As in the overview of existing meta-analyses, also a variety of ways can be distinguished in which forest biodiversity and ecosystem services are detailed as attributes in the policy scenarios. Number of (endangered) species is one of the most common representations of biodiversity. In other cases, biodiversity levels are introduced. Only one study was found where the attributes only consisted of ecosystem services (Mogas et al. 2006). In a recent study focusing on public preferences and valuation of different types of nature compensation projects with the aim to estimate a value transfer function for Belgium (Liekens et al. forthcoming), forest was valued against 5 other different biomes to account for possible substitution effects, including agricultural land, marshland, natural grasslands, open water, and heath land. Biodiversity was, however, only presented in this study in two simple levels (low with few common species or high including endangered species). In those studies where biodiversity was included alongside ecosystem services, these services usually referred to cultural ecosystem services such as recreation, scenery, aesthetics and information and communication.

Table 5: Overview of forest attributes used in existing choice experiments

No.	Authors	Country	Attribute 1	Attribute 2	Attribute 3	Attribute 4	Attribute 5	Attribute 6	Payment vehicle
1	Hanley et al. 1998	UK	Felling scheme	Shape	Species mix				Tax
2	Mallaawarachi et al. 2001	Australia	Area of teatree woodlands	area of vegetation along rivers and in wetlands	regional income from cane production				Annual levy on land rate
3	Holmes and Boyle 2003	USA	Forest road density	Dead trees after harvest	Live trees after harvest	Maximum size of harvest area available for harvesting	Width of riparian buffers	Slash disposal	One-time tax increase
4	Lehtonen et al. 2003	Finland	Information and education	Conservation contracts	Conservation areas	Biotopes at favorable levels of conservation	Number of endangered species		Increases in annual income tax 2003–2012
5	Xu et al., 2003	USA	Management strategy	Biodiversity	Aesthetics	Rural forest job losses			Additional costs
6	Garber- Yonts et al., 2004	USA	Salmon habitat	Endangered species protection	Forest age management	Biodiversity reserves			Price a household would have to pay
7	Watson et al., 2004	Canada	Protected areas in percent of total region	Age of stands	Recreation access	Biodiversity levels			Changes in taxes
8	Horne et al., 2005	Finland	Species richness at each site	Average species richness	Variance of species richness	Scenery at each site			Change in municipal taxes
9	Naidoo and Adamowicz, 2005	Uganda	Number of bird species seen	Travel time	Visit part of tour?	Lodging facilities	Landscape features	Chance of seeing large wildlife	Entrance fee

No.	Authors	Country	Attribute 1	Attribute 2	Attribute 3	Attribute 4	Attribute 5	Attribute 6	Payment vehicle
10	Bienabe and Hearne, 2006	Costa Rica	Number of conservation-focused zones	Number of scenic beauty/access-focused zones					Payment through airport taxes (tourists) or municipal taxes (Costa Ricans)
11	Horne, 2006	Finland	Initiator of the contract	Restrictions on forest use	Duration of contract	Cancellation policy			Compensation/ha/year
12	Cristie et al. 2006	England	Familiar species of wildlife	Rare, unfamiliar species of wildlife	Habitat quality	Ecosystem process			Annual tax increase
13	Mogas et al. 2006	Spain	Picnicking allowed in new forest	Driving allowed	Mushrooms	CO2	Erosion		Afforestation cost per person per year
14	Nielsen et al., 2007	Denmark	Species composition	Tree height structure	Standing and fallen dead trees				Increase in annual tax payment per household
15	Wang et al. 2007	China	Sandstorm days per year	Landscape (vegetation cover)	Water quality (billion tons of annual sediment discharge)	Plant species present			Payment per annum
16	Liekens et al. (2012)	Belgium	Nature type	Area size	Biodiversity level	Distance from respondent home	Adjacent land use	Public access	Annual tax increase

6.8 Experience with economic valuation in the POLICYMIX case studies

In all PolicyMix case studies, the WP4 guidelines have been used to a larger or smaller extent. In three case studies, Norway, Germany and the Netherlands, an actual valuation study has been carried out (and one is under way in Portugal). In most other case studies, WP4 guidelines have been used to structure the comparison between costs and benefits of policy instruments, mostly using some form of cost-effectiveness analysis.

In the Norwegian case study, a contingent valuation study was performed in the context of a cost-benefit analysis of an extension of conservation of old-growth forests. The study focused on the valuation of cultural services derived over a 50 year time horizon. The results showed that an increase of conserved area passes the cost-benefit criterion for each of the scenarios considered.

In the German and additional Dutch case study, a choice experiment was carried out to assess the willingness and motivation of land owners for afforestation in a PES-type arrangement. Specific attention was paid to issues related to contract design and the institutional-economic terms and conditions needed to be in place to increase farmers' uptake of existing agro-forestry schemes. Preliminary results show that a combination of morivational and economic factors (compensation levels) play a significant role in explaining landowners' decision to participate in these existing schemes (or not).

6.9 Relevant links and further sources of information

Further sources of information on economic valuation methods are found in:

Economic valuation generally

1. Pearce, D., D. Moran, D. Biller (2002), Handbook of Biodiversity Valuation, a Guide for Policy Makers, OECD Publications, OECD, Paris.
2. Freeman, A.M. III (2003), The measurement of environmental and resource values, Theory and methods, Resources for the Future, Washington, D.C.
3. World Bank (2003), A Review of the Valuation of Environmental Costs and Benefits in World Bank Projects, Paper no. 94, Environment Department Papers, World Bank.
4. World Bank (2004), Assessing the Economic Value of Ecosystem conservation, Washington, DC: World Bank.
5. National Research Council (2004), Valuing Ecosystem Services, Toward Better Environmental Decision-Making, National Academy Press, Washington, DC.
6. Kontoleon, A., Pascual, U., and Swanson, T. (2007). Biodiversity Economics. Cambridge: Cambridge University Press.
7. http://www.ecosystemvaluation.org/

8. http://www.naturalcapitalproject.org/InVEST.html

Nonmarket valuation methods

Stated preference methods

9. Mitchell, R.C., R.T. Carson (1989), Using Surveys to Value Public Goods, The Contingent Valuation Method, Washington, DC: Resources for the Future.
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Revealed preference methods

16. Ward, F.A., D. Beal (2000), Valuing nature with travel cost models, A manual. Edward Elgar Publishing, Cheltenham, UK.

Stated and revealed preference methods

17. Champ, P., K.J. Boyle, T.C. Brown, (eds.) (2003), A primer on nonmarket valuation, Kluwer Academic Publishers, Dordrecht, Netherlands.
18. Bennett, J. (2011), The international handbook on nonmarket valuation, Edward Elgar Publishing, Cheltenham, UK.

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