What Science Can Tell Us

# The Provision of Forest Ecosystem Services

## Volume I: Quantifying and valuing non-marketed ecosystem services

Bo Jellesmark Thorsen, Robert Mavsar, Liisa Tyrväinen, Irina Prokofieva and Anne Stenger (editors)

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**Bo Jellesmark Thorsen, Robert Mavsar, Liisa Tyrväinen, Irina Prokofieva and Anne Stenger (editors)**

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# 1.

# Executive overview

**Bo Jellesmark Thorsen and Sven Wunder**

The forests of Europe provide numerous goods and services for the benefit of Europe's citizens. Wood is the most prominent, but game, cork, mushrooms and greenery are also traded in significant volumes. However, many forest goods and especially services are not marketed, but are still of great value. Forests play an essential role in water resource management from local to regional levels. Forests are crucial for the preservation of biodiversity; many threatened terrestrial species depend upon forest habitats for survival. Forests' ability to sequester and store carbon is crucial to the mitigation of climate change. In addition, forests form an important part of landscape amenities, cultural heritage, and are of great recreational value.

The term 'ecosystem services' or the slightly narrower sister term 'environmental services' has been used since the 1990s as an umbrella term for various goods, services and functions. A much cited study by Costanza et al. (1997) even attempted to assess the total value of the world's ecosystem services and natural capital; the number found in fact exceeded the total gross national product of the world. Yet, the study has been widely criticized for putting an absolute value on something that cannot be fully substituted, thus extrapolating economic valuation beyond its meaning: a tool for evaluating well-defined choices of ecosystem management and protection. Hence, the single bottom-line number still leaves important questions unanswered: *How far can ecosystems be exploited, modified, and degraded before net welfare losses are registered? Can societies gain from enhancing the protection of biodiversity and habitats and the provision of ecosystem services? What methods and policy measures may be used for determining and pursuing the answer to these questions?*

Several ongoing international agreements and policy developments relate to the latter question, including the Convention on Biodiversity and the newly started Intergovernmental Platform on Biodiversity and Ecosystem Services), the European Union supported work on The Economics of Ecosystem and Biodiversity (TEEB 2010a, b) leading to the current EU MAES process, focusing on the mapping and assessment of ecosystem services. In direct and indirect ways also the post-Kyoto Conference of the Parties (COP) process has address also this question, e.g. in the discussions of how to reduce emissions from deforestation and forest degradation.

To pursue the answers to these more crucial questions, science needs to provide several pieces of knowledge, which relate to underlying policy relevant questions. Based on new analyses from a larger EU-project[1] on forest ecosystem services, this and the accompanying volume provide new insights and examples needed to answer questions such as:

*What will be the value for society of a specific enhancement of ecosystem service provisions in a specific spatial context, and how are benefits distributed?* This calls for the further development of environmental valuation techniques and analyses that allow us to estimate also values of non-marketed ecosystem services. Many of these are best characterized as externalities, in the sense that the positive or negative impacts determined by the landowners' management decisions fall on other off-land agents.

*What will be the costs of enhancing ecosystem provisions in specific contexts?* Enhancing the provision of e.g. recreational opportunities may come at costs in terms of lost forest production but also costs relevant for society in terms of reductions in other ecosystem services. These needs to be assessed and again the variation across different contexts and owner types are of interest for policy makers.

*What will be suitable policy instruments for society to balance costs and benefits in the best possible way in each context?* To address the overall issue of reaching a sustainable balance between use and protection of our natural ecosystems, we need intelligent choice of policies. We present new insights into the view that both the public and forest owners have on the design of such instruments.

We highlight in Box 1 some of the many new insights and lessons learned from our research, that provide new, improved and context relevant answers to the overall questions.

## A closer look at the challenges

A central issue in society's pursuit of the best provision of ecosystem services and notably those that are externalities is that the private landowners so often crucial to their provision are not rewarded for the provision through the markets. This means that provision will be too low relative in particular to the provision of marketed goods like wood, hunting rights etc. To remedy, this society may put in place rules or other mechanisms to direct or encourage the landowners to change behaviour in ways that enhance aggregate welfare.

Figure 1 illustrates how this may be resolved. Assume a forest owner's privately most profitable land-use option is to intensify the management of a forest area, currently yielding Q4 in its extensive use, providing the superior gain Q1. The forest owner gains Q4 – Q1 from this. Assume further that this change in management would reduce local biodiversity through habitat loss, carbon storage through tree loss, water quality through more erosion, and recreational values through diminished landscape beauty. We use non-market valuation techniques to quantify society's combined losses of ecosystem services values at Q2. The large potential loss, however, may jointly induce service users and beneficiaries (perhaps represented by the state) to offer the forest owner payments for the environmental services (PES) equalling Q3 – sufficiently small to not exceed externality values Q2, and sufficiently large to at least compensate the landowner for the gain he would forgo from not intensifying his management (Q1–Q4).

In parenthesis, we could imagine other incentives to compensate landowners, whenever these are entitled to freely make resource-use decisions independent of externalities

1 The volumes draw upon newer research findings and in particular uses new case insights from the EU FP7 project NEWFOREX (243950), completed May 2014.

### Box 1. Lessons learnt from recent research.

**Quantification of and goal setting for non-marketed forest ecosystem services (ES)**

- Any policy targeting ES should have clear and measurable goals for ES quantities at least for two reasons: To ensure that what is being delivered is what has value and to allow society to monitor the efficiency of policies
- In goal setting, it should be remembered that any policy will likely affect several ES and therefore multiple policies may be needed for balance

**The valuation of non-marketed forest ecosystem services (ES)**

- Using improved methods we add documentation for the impressive value of non-marketed forest ecosystem services – yet we argue that to make valuation studies policy relevant, focus should turn away from total economic values to value distributions
- Environmental policies have distributional effects: Some people win more than others – and others again may lose. We demonstrate with case studies that these differences are not trivial and likely to be highly policy relevant
- Identifying who values ES how much can inform policy instrument design in order to gain legitimacy and direct costs to where values are harvested.

**The cost of provision for non-marketed forest ecosystem services (ES)**

- We demonstrate the benefits of applying multiple methods for assessing the cost of provision – capital budgeting techniques widely used can be further informed by methods taking forest owner perceptions into account
- We document that European private forest owners are generally positive towards the provision of ES from their forests
- We document how differences in forest owner objectives spill over over to major heterogeneity their perceived cost of providing further ecosystem services. This opens up options for improved cost efficient policy designs

**Economic instruments for non-marketed forest ecosystem services (ES)**

- We demonstrate that many formal aspects of contract matter and that loss of decision right is costly, thus instruments should be designed to limit these where possible and carefully consider aspects like exit options, time frame etc
- We document that participation rates in voluntary economic instrumentss increase when transactions costs can be controlled, e.g. largers forest properties, higher educated and forest owners with experience from other instruments are more likely to enter a new instrument
- We document that forest extension companies can be instrumental in reducing transactions costs and stimulate participation from owners who face steep transaction costs
- We find that ES targeted instruments are more likely to attract forest owners if the are aligned with forest owner values – for example instruments requiring action (infrastructure, establishing new nature, restoration) are seen more positive than instruments requiring inaction (passive conservation) – policy instruments can be designed to benefit from this
- We document that the majority citizens of several European countries support the view that cost of ES provision should in general be carried by society or identified users directly – and not the forest owners. This shows widespread public support for economic instruments.

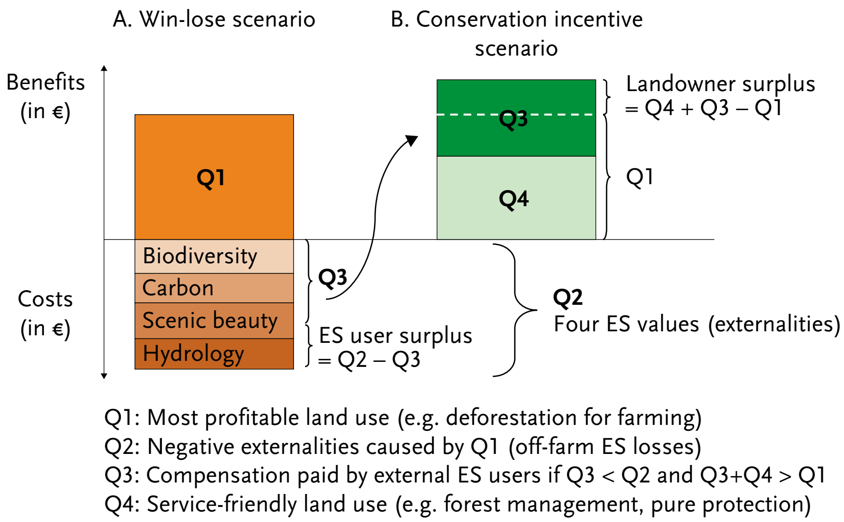


**Figure 1.** Externalities in ecosystem services provision. Modified from Pagiola and Platais (2007).

(*provider gets principle*). Alternatively, the government might instead choose to hold the landowner responsible for the externalities they cause, and impose a tax on them equalling Q3 (*polluter pays principle*), or simply prohibit the management change with appropriate sanctioning. Whether incentives, disincentives, or regulatory policies are more appropriate will depend on legality, *de facto* entitlements, as well as the political economy context of natural resource management.

How do we specifically value Q3? Non-market valuation methods have targeted various forest ecosystem services, e.g. the value of forest proximity, access rights and recreation, the value of forest biodiversity protection and wildlife and the value of forests' potential for carbon sequestration. It is a general finding that the demand for, and awareness and value of these goods and services are on the increase in many European contexts. Yet, methods for assessing in an integrated way these often jointly produced values has recently seen much further development, allowing us not only to assess values of ecosystem services independently, but also jointly when produced as a part of the same forest management practice.

Environmental valuation studies only recently started addressing the fact that environmental policy – like any other policy – also has distributional impacts.

For society, it may not be enough to know the aggregate welfare gain or loss from a change in an ecosystem service. It is also important to consider how such gains and losses are distributed: Who gains, who loses, and how much?

Assessments of distributional impacts are integral to all policy arenas, yet in environmental policies and valuation, it remains understudied even in advanced countries. This is particularly true when we study ecosystem services of a public good nature, where

ES users cannot (easily) be excluded, and when values accrue trans-nationally (e.g. biodiversity, carbon).

In many European countries, much forest area is privately owned, and very often by many small-scale proprietors. For policies targeting ecosystem services provision, it is challenging that these predominantly are externalities of forest management that provide no economic return to the forest owner. If forest owners are not compensated for the costs incurred in producing the value of such externalities, they may manage their forests in ways where external benefits are being provided in lower amounts than socially desirable (see Figure 1). Therefore, there may be significant potentials for welfare gains, and it is an important research challenge firstly, to assess when adequate incentives and other regulation approaches can be developed to materialize these gains by enhancing ecosystem service provision, and secondly, to then customize them to the variations in forests, forest owners, and socioeconomic contexts around Europe. Still, much forest also remains in public ownership, which may ease ecosystem service provision. Many intermediate forms of semi-public ownership exist, which may require tailored instruments to achieve better outcomes.

To enable an efficient functioning of novel policy instruments, it is important to understand how different framework conditions influence the functioning of such instruments, as well as their legitimacy and acceptability among the public as well as forest owners. In Europe the use of economic instruments in environmental policy often relies on large programmes with public funding for supporting environmentally friendly land-use decisions at landowner levels in the Common Agricultural Policy (CAP). Public funds are scarce, so cost efficiency is a major concern, which in turn also relates to the perceived legitimacy of such instruments. One aspect to consider here is that costs of enhancing ecosystem service provision vary across forest owners, and cost efficiency therefore requires differentiated tools.

Thus, it is a challenge for research to elaborate methods for assessing cost-of-provision that can inform policy design and ensure cost effective implementation across forest owners.

## Presenting new results from recent research

In these two "What Science Can Tell Us" volumes, an international group of researchers summarizes and presents in a number of short, focused chapters – and using a set of supporting case studies – the complex pan-European world of forest ecosystem services with novel findings and insights that shed new light on several of the above questions. Most case studies come from the EU FP7 project NEWFOREX, which ended in May 2014. The NEWFOREX project is one of several larger EU projects addressing ecosystem services, it is one of the only projects concentrating on economics and policy and it is the only project concentrating on forestry[2]. The empirical fundament

2 Related projects of interest include POLICYMIX (http://policymix.nina.no/), OPERA (http://operas-project.eu/) and OPENNESS (http://www.openness-project.eu/).

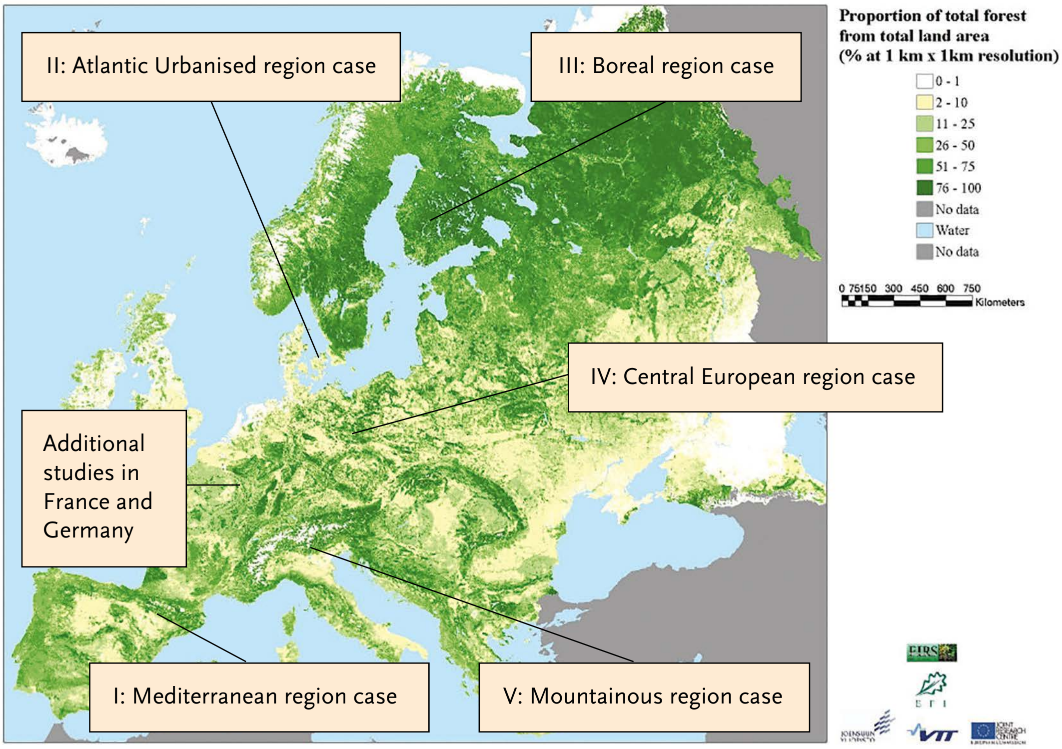


**Figure 2.** The five European case studies in NEWFOREX, and the supporting French and German forest owner studies. The developing country case in Brazil is not shown.

of NEWFOREX contained five regional case studies in Europe, and additional targeted studies in France and Germany (see Figure 2). Finally, a tropical case study from the Brazilian Amazon was included for comparative, extra-European perspectives in particular on carbon sequestration.

The structure of the volumes reflects the above challenges: In this the first volume, we discuss the concept of ecosystem services and how to adequately quantify them in ways that are suitable for linking them to the benefit people derive, to the management measures that may change ecosystem service provision and through that to the costs landowners and society at large may carry when changing management. We focus on a selected set of ecosystem services to that end. Furthermore, we discuss the challenges and potential in obtaining monetary measures of value for non-marketed forest ecosystem services, externalities. To illustrate we provide a series of short applied chapters providing examples of how to measure the value of the selected set of ecosystem services.

In the second volume we address the two remaining questions. We discuss how the provision of ecosystem services can be enhanced by changing forest management and address three important issues: first, the definition, measurement and quantification of the management changes in terms of inputs and/or outcomes (e.g. ecosystem services); second, the assessment of the related costs of provision arising from changes in forest management; and third, the use and design of in particular economic policy instruments for enhancing the provision of forest ecosystem services.

## What is yet to be learned?

Research on sustainable management of ecosystem services from forests and all other kinds of ecosystems and biomes are increasing in volume these years with the renewed focus on the value of biodiversity and ecosystem services in combination with the increasing pressure on natural resource for renewable materials.

Yet, much more remains to be learned, and the largest gaps in our knowledge often remain natural science in nature: Understanding the dynamics across various ecosystem services at various spatial scales, understanding the role of various species in ecosystem functioning and ecosystem service stability and quality, understanding and predicting the likely impacts of climate change on habitat and ecosystem development and in turn ecosystem service provision.

Getting to grips better with the answer to these and many similar gaps in our knowledge will pave the way for applying in even greater detail and with greater precision, several of the current methods for economic and policy analysis for ecosystem service provision and management presented in these volumes. However, there are also numerous open questions in a social science perspective. Perhaps first is the question how changing pattern of land ownership and land owner objectives may affect management decisions and ecosystem service provision? And in connection to this, how can policy instruments for enhanced provision be designed to take into account not only spatial variation in ecosystem service values and supply potentials, but also the heterogeneity of land owners? Also, we are short of empirical research addressing forest owners' likely decision strategies in the face of climate change and analyses of how this may affect ecosystem service provision and stability.

## Recommended reading

Costanza, R., d'Arge, R., de Groot, R. Farber, S., Grasso, M., Hannon, B., Limburg, K., Naeem, S., O'Neill, R.V., Paruelo, J,. Raskin, R. G., Sutton, P. and van den Belt, M. 1997. The value of the world's ecosystem services and natural capital. Nature 387: 253–260.

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# 2.

# Ecosystem services and their quantification

## 2.1 What are ecosystem services?

**Sven Wunder and Bo Jellesmark Thorsen**

Ecosystem services have recently become a key concept in understanding and conceptualizing the way humans interact with the natural environment. They represent what can be broadly understood as the multitude of natural resources and processes that humans benefit from. It is thus by nature an anthropocentric, utilitarian concept, in addition to which we may also consider nature's own right to exist and thrive. Note that the about equally popular term "environmental services" is largely used as a synonym, though perhaps sometimes in a more separable and less landscape-level holistic way. The Millennium Ecosystem Assessment (MEA) included four basic categories of ecosystem services:

1) Supporting services: basic services deemed necessary for the production of all other ecosystem services (such as nutrient and seed dispersal from forests);
2) Provisioning services (delivering tangible products, such as from forests foodstuff, fibres, and fuelwood);
3) Regulating services (for instance, for forests hydrological regulation or climate change mitigation);
4) Cultural services (such as forest recreational or benefits or their role in religion and environmental education).

The Millennium Ecosystem Assessment (MEA), a four-year study involving more than 1,300 scientists worldwide, popularized the ecosystem services approach, and showed how humans depend vitally on the different types of services provided.

By altering the structure and functions of ecosystems, humans will tend to impact type and size of service flows that promote human well-being, thus also linking ecosystem services to economic development and land-use dynamics. In a world where resource allocations are increasingly linked to evidence-based societal benefits, the idea of featuring

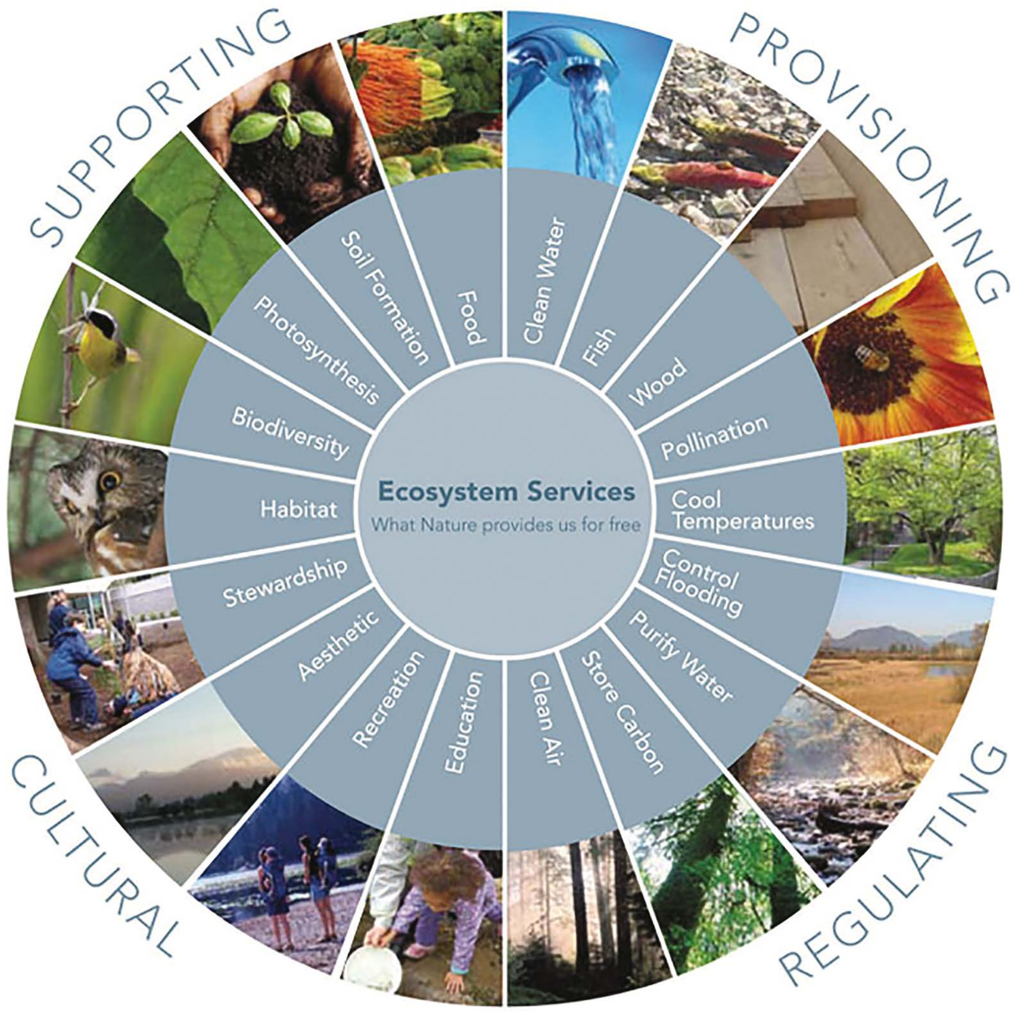


**Figure 3.** The Millennium Ecosystem Assessment (MEA) organizes 'ecosystem services' into four broad categories. Source: Metro Vancouver.

verifiable and at best quantifiable benefit flows has gained good traction, as it makes explicit a link to poverty alleviation, equity, and human welfare. Many conservation organizations and environmental decision-makers have thus restructured their interventions around the concept of ecosystem services, linking ecosystem management options explicitly to tangible stakeholder interests.

The ecosystem services definition adopted by MEA was a particularly broad one, and has correspondingly also triggered significant critique. In the recent influential TEEB (The Economics of Ecosystems and Biodiversity) exercise, the MEA's overarching "supporting services" have been relabelled as "habitat or supporting services", struggling in particular with problems of service double-counting. The "cultural services" concept has been criticized for including symbolic, interpreted landscape values that are often only ambiguously related to ecological function and by their nature basically unique and incomparable across sites. Finally, the "provisioning services" concept arguably blurs a longstanding distinction between products and services, or between tangibles and intangibles, with different ownership, user right and governance mechanisms.

Table 1. Illustrating excludability, rivalry and classification of forest ecosystem services (adapted from Buyers 2008).

| Excludability<br><br>Degree of rivalry | Excludable<br>(can limit access) | Non-excludable<br>(cannot or do not limit access) |
|---|---|---|
| Rival<br>(Consumption by one reduces options for others) | Private Good<br>Examples: Timber, sometimes game when rights are well-defined and private | Common Pool or Open Access Resource<br>Examples: Forest fruits, mushroom, when not subject to private ownership. Recreation with congestion effects |
| Non-rival<br>(Consumption by one has no effect on the consumption option of others) | Club Good<br>Examples: Watershed protection services, ecotourism in protected areas, hunting clubs | Public Good<br>Examples: Forest biodiversity non-use values, climate change mitigation, water system regulations, fire prevention |

Two concepts have proven useful when thinking about the properties of various forms of goods, service and resources, and that are those of 'rivalry' and 'excludability'.

A good is subject to rivalry, when it cannot be simultaneously consumed (enjoyed) by more than one individual in a meaningful way. As we can see in Table 1, "provisioning services" (or better, ecosystem products like timber, fuels etc.) all fall into the category of rival goods, which gives them completely different propriety, usage and management characteristics than the family of non-rival intangible services. A good or service is non-rival when the consumption by one individual does not inhibit the simultaneous consumption of the good by others. Good examples are a nice view, carbon emission reductions and similar, and in fact many MEA denominated service categories are largely non-rival. The second concept is that of excludability. This refers to the ability (by law, rules, physical or natural barriers) of one or more individuals to exclude others from consumption of a given good, service or resource. Table 1 illustrates these concepts in a simple form, but note the concepts are not categorical. Rather any good could be placed somewhere on the continuous axes from complete rivalry to non-rival and complete excludability to no excludability. Eventually, these differences have important consequences both for the valuation and the management of products and services.

In these two volumes we have focused on in particular on four key externalities: watershed protection values, biodiversity protection values, carbon benefits and recreation.

All of these are rarely placed in the private good category above, they may in some cases (e.g. in a watershed) have the characteristics of a club good, but in general they are either public goods or have common pool characteristics. However, we also discuss examples of goods and services that sometimes belong in the private good category, example when rights to mushroom picking are well defined and protectable, and sometimes not.

While the MEA's extremely broad classification can serve political purposes of demonstrating the existence of ample natural values, for all practical purposes of managing ecosystem goods and service, a focus on the "regulating services" (corresponding to the pre-MEA definition of the "ecosystem services" term) may be preferable. To become a fully operational planning tool, important spatial trade-offs have to be recognized, not only of extracting tangible products versus maintaining intangible services, but also analysing complex correlations between different intangible services, representing heterogeneous landscape mixes of service synergies and trade-offs. Tools have been developed with focus on spatial overlays of service types with environmental threat and conservation costs and on the spatial integration of the demand and supply sides of the service economy. In this sense, while the definition of ecosystem services may have been clouded somewhat by well-intentioned efforts to broaden their scope, for their core regulatory components some important practical steps have already been taken to mainstream them into forestry and conservation planning.

## Key messages

1. The biophysical aspects and the socioeconomic and legal context together determine the nature of ecosystem services from a policy viewpoint.
2. Major forest ecosystem services are often non-marketed public goods or common pool resources.
3. An adequate provision of forest ecosystem services requires adequate institutions and policies to support beneficiaries and suppliers in efforts to improve provision.

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# 2.2

# Quantifying water externalities from forests

**Sven Wunder and Bo Jellesmark Thorsen**

Watersheds and the functioning of the hydrological cycles are crucial not only for the natural ecosystems, but also provide crucial ecosystem services for mankind. Forests as compared to other vegetation covers influence hydrological cycles. Hence, the management or mismanagement of forests can affect the hydrological outcomes that humans want ecosystems to provide.

One fundamental characteristic of forests relative to other vegetation covers is a higher evapotranspiration, resulting from the large size and biomass of trees and their larger vegetative surface.

Thus, forests tend overall to consume more water than most other vegetation types, and may reduce river runoff correspondingly – a fact that greatly affects the role of forests in providing water-related ecosystem services. It should be stressed; however, that there is considerable variation in evapotranspiration across forest types, e.g. according to dominant tree species, and therefore forest management can also directly impact the provisioning of water-related services. Yet, in some rare cases forests can also work as water-capturing towers intercepting clouds, mists, and condensation.

The second major effect is that forests and trees are favourable to permeable soil structures that can be penetrated and store and filter water, rather than causing immediate and potentially destabilizing surface runoff. The third principal effect is that forests are good at protecting soils from eroding, especially on sloped terrain, which can have major effects on downstream users. Finally, the presence or not of forests has impacts on micro climates, and as recent research indicates, probably also the climate of macro regions, implying that forests could function as atmospheric "water pumps" without which the areas of the Amazon or the Congo Basin could be deserts.

These combined vegetation and soil impacts can thus also enhance a series of hydrological sub-services: seasonal stream flow quantity, control of its variability and quality (including sediments, nutrients, and pollutants) and risk management (including flood and landslide prevention).

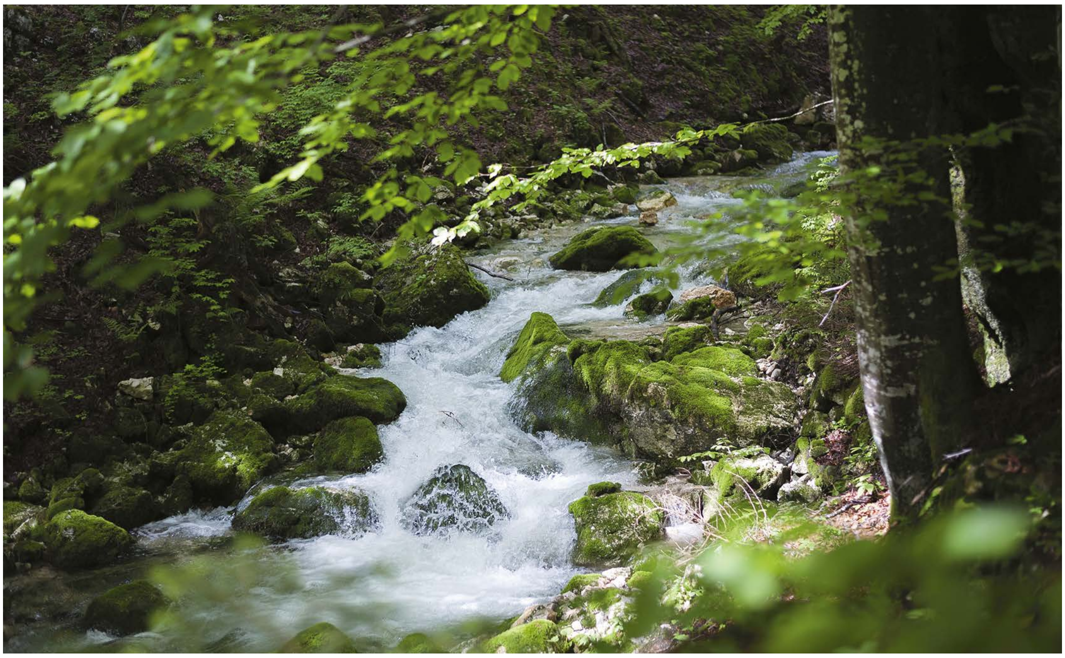

Figure 4. Intact or well-managed natural forest cover can regulate stream flow, including regulation of seasonal flows, providing soil protection by avoiding erosion and stream sediment loads. Photo: Jaka Zvan, fotolia.com.

## Hydrological ecosystem services from forest

The many hydrological functions that forest ecosystems can enhance sound impressive, but many of the underlying linkages between land-cover/use and services provided are spatially highly context-specific, both from the service supply side (e.g. presence of sloped terrain and fragile soils causing high erosion and flooding risks) and from a value point of view from the demand side (e.g. presence of large cities downstream depending on these water services). Here we briefly describe a few of these services in broader terms to allow for insights into the scientific state-of-the-art.

Forests' ability to intercept rain and snow, improve soil structures, and hence reduce run-off represents an important ecosystem services in watersheds where erosion, floods or avalanches implies costs, such as risks of landslides or avalanches damaging infrastructure or productive croplands, residential property, environment and human health of downstream floods affecting both rural and urban areas or the costs associated with sedimentation of rivers and lakes, reducing quality and values of these for many purposes, including e.g. to avoid the siltation of hydropower dams and drinking water reservoirs.

Intact or well-managed natural forest cover can regulate stream flow, including regulation of seasonal flows, providing soil protection by avoiding erosion and stream sediment loads.

Major reforestation programmes, such as in China, have been implemented with the primary aim of mitigating flood risks. However, reforestation is unlikely to reduce flood

Table 2. Examples of how changes in forest characteristics affect changes in ecosystem functions and services (Δ denotes change).

| Change in characteristics | ... affect Ecosystem Function | .. and Ecosystem Services |
|---|---|---|
| Change in forest cover | Δ Erosion and run-off control | Δ Sedimentation of streams, lakes and dams<br>Δ Risks of floods and avalanches |
| Change in forest cover | Δ Water capture from air | Δ Water supply for household and industry use<br>Δ Water available for other ecosystems |
| Change in forest species | Δ Evapotranspiration and groundwater recharge | Δ Groundwater available for society |
| Change in harvest regime | Δ In nutrient losses with leaching | Δ Groundwater and run-off quality |

risk in the same way as old-growth forest because the recovery of degraded soils can take decades, implying a medium-run irreversibility in the consequences of forest management. Conversely, intact natural vegetation cover *per se* is no guarantee that floods or landslides will not occur, especially in large scale watersheds and under extreme weather events. Nevertheless, their frequency will be less with intact vegetation than is usually observed after conversion, especially in smaller-scale watersheds. In regions where groundwater is the major source of drinking water, the role of forests for aquifer recharge quantity and quality can be important. Aquifer recharge could, depending on soil properties, be lower under forests than, say, under agricultural crops, but it will usually contain much lower concentrations of nutrients ($NO_x$) and pesticides. Thus, afforestation may reduce quantity somewhat, but increase water quality, implying trade-offs between different attributes of hydrological services.

There are significant differences in the net groundwater recharge between coniferous and broadleaved forests and forest management may enhance or reduce the quantity of groundwater recharge.

Forests' regulating impacts on water courses also play a role in some fairly rare ecosystems and cases where, e.g. the so-called cloud forests, function as protectors and providers of stable clean water, as such forests intercept ground clouds or serve as mist, fog and night dew interceptors concentrating and collecting water on the surface of the trees' foliage and leading much of this into the ground, feeding other vegetation types, groundwater and streams. In such regions, where water from other sources are perhaps in shortage, forest cover may have direct positive impact also on available water resources for other ecosystems as well as household uses.

To be able to evaluate the desirability of a change in the provision of water related ecosystem services, it is necessary to be able to identify the link between forest management measures and the resulting effects on the ecosystem functions and in turn the affected ecosystem services, which often may be manifold. In Table 2, we provide some tentative illustrative examples of links from changes in forest characteristics that are sensitive to management decisions over the ecosystem functions to the actual services provided.

Table 3. Examples of various changes in ecosystem functions and services, and what relevant measures may be (Δ denotes change).

| Ecosystem function | Ecosystem Services | Measure of service |
|---|---|---|
| Δ Erosion and run-off control | Δ Sedimentation of streams, lakes and dams | Δ tons of sediment |
| Δ Erosion and run-off control | Δ Risks of floods and avalanches | Δ assessed risks and values at risk |
| Δ Water capture from air | Δ Water supply for household and industry use | Δ $m^3$ available |
| Δ Water capture from air | Δ Water available for other ecosystems | Δ $m^3$ available |
| Δ Evapotranspiration and groundwater recharge | Δ Groundwater available for society | Δ $m^3$ available |
| Δ In nutrient losses with leaching | Δ Groundwater and run-off quality | Δ in concentrations per litre |

## Quantifying water services for environmental valuation

While links such as those illustrated in Table 2 represent likely effects, they are not *per se* sufficient to decide on the desirability of management for one hydrological service over another. They can form a basis for assessing costs of provision.

However, to assess benefits of water related forest ecosystem services, we need to quantify the latter in ways that allow us to estimate sensible values of the changes in quantities or qualities of the ecosystem services.

It is a complex task in many cases to decide on relevant units and measures of the ecosystem service change and not least to actually measure and quantify these changes in given cases. What are the changes in avalanche or land slide risks from changes in forest covers in various places of a mountainous region? How can we measure and express the effects of changes in forest management for groundwater quality and quantity in ways that people in general can relate to?

Table 3 illustrates in a simplified fashion how various relevant measures may be formulated and related to changes in ecosystem functions and services. For simplicity, we ignore per area or period parts of the units of measurement. Linking Table 3 and Table 2, we can deduce the pathway from, e.g. a desired change in the amount of groundwater recharge in an urban region depending on groundwater for drinking water, and back to the forest management change that can bring it about. This is exactly the case considered in the Atlantic case study of NEWFOREX, situated in the greater Copenhagen area of Denmark. The capital city pulls such a heavy draw on the groundwater resources to affect groundwater levels and water levels of streams and lakes in summer periods in almost all parts of Zealand.

Switching current coniferous forests to broadleaves on the island of Zealand can bring about an additional groundwater recharge of some 20–40 million $m^3$ annually across the island. This corresponds to the consumption of some 250–300,000 households.

Table 3 also shows that changes in ecosystem functions may impact several ecosystem services, and that these ecosystem services may again have several dimensions of relevance (in terms of quantity, quality and end-use). Therefore, discipline needs to be applied when valuations are made, and in particular when values are aggregated. These aspects along with often quite ambiguous definitions of ecosystem services implies a risk both of double counting and of under counting the values of ecosystem services, including hydrological ecosystem services from forests.

In a later chapter we present an example of how the link between forest management changes (species change from conifers to broadleaves) and groundwater recharge can form the basis for the valuation of increased groundwater provision for drinking water by potential end-users.

## Key messages

- Water related externalities of forest use are complex and comes in a variety of forms.
- Forest management can affect groundwater recharge, quality and surface runoff and erosion significantly – with potentially large gains or losses for society.
- Biophysical models linking site and forest characteristics to management actions allow us to quantify these effects and hence assign values to management impacts.

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2.3

# Forests and carbon externalities

**Sven Wunder and Bo Jellesmark Thorsen**

As stressed by Stern (2006), the climate change phenomenon represents the perhaps largest failure ever of mankind's inability to account for large scale market failures. The historic emissions of $CO_2$ relies not only on the industrial use of fossil fuels, but to a significant extend originates also from past and current deforestation activities. Thus, forests represent a potential source of immense emissions, but for the same reasons actually also represents a potential sink of significant scale.

Compared to most other vegetation covers, trees and forests are particularly rich in biomass and carbon content, stored partially in the living wood of tree trunks, but also in roots, leaves, deadwood, forest floors with debris, forest soils organic matter, etc. – and eventually also in wooden products taken out of the forest. Hence, to maintain and improve forest cover with high carbon content has been outlined strategically as one of the quickest and cheapest ways of mitigating climate change originating from the anthropogenic accumulation of greenhouse gas (GHG) emissions in the atmosphere (Stern 2006).

In addition to climate change mitigation, forests can also play a role in the adaptation to climate change, i.e. in making sure humans will be able to cope with the consequences of that part of climate change that we will prove unable to mitigate. In part, these potentials lie in the non-carbon ecosystem services that forests produce, such as hydrological protection and safeguarding of microclimates (see last section). In part, this refers also to the income generation potentials of forest products in the face of climatic stresses on other sectors, such as agriculture – provided that forest ecosystems themselves can maintain their productivity in the face of climatic change (Guariguata et al. 2008).

In this section, we will nevertheless concentrate on forests' mitigation potential. This relates to forests' carbon sequestration, and represents a pure externality and public good that landowners usually do not directly benefit from, absent compensation mechanisms. However, unlike forest hydrological services, carbon services are not spatially specific: GHG mitigation can be achieved anywhere in the atmosphere through forest carbon sequestration in any of the world's forest. Thus, the beneficiaries are by definition all global inhabitants benefiting from avoided climate change. Furthermore, carbon services are much more homogeneous than the large variety of hydrological services: the former can all be converted to a single measurement unit: $CO_2e$ (carbon dioxide equivalents).

## Carbon ecosystem services from forests

Forests provide mitigation options under what the IPCC has termed Land Use and Land Use Change and Forestry (LULUCF) (http://unfccc.int/essential_background/glossary/items/3666.php#L), which also includes e.g. changes in various agricultural practices.

Table 4. Examples of how changes in forest management will affect carbon balances (Δ denotes change).

| Change in management | ... effect on land | .. and carbon |
|---|---|---|
| Avoiding deforestation | Δ in forest conversion for other land uses | Δ Carbon stock on land in focus maintained<br>Δ Avoided carbon emission flows from alternative land use |
| Avoiding forest degradation | Δ in selective (hit-and-run) logging of trees and similar practices | Δ Higher carbon stock on land maintained<br>Δ Avoided loss of carbon from wood consumption process |
| Re- and afforestation | Δ Increased forest land cover | Δ Growing carbon stock for decades on land in focus<br>Δ Possible carbon loss/gain from stopping alternative land-use emission flows |
| Forest (carbon) enhancement d) | Δ Management activities, e.g. species choice and thinning | Δ Carbon stocking and production of wood that may replace alternative materials and fuels with associated emission flows |

The hoped-for climate change mitigation potentials from forests can logically be realized in a variety of ways (cf. Table 4).

The carbon restocking options c) and d) in Table 4 are what have been dominating over the last century in the Northern Hemisphere, due to both forest regrowth on abandoned marginal agricultural lands, depositions of nitrogen from agriculture and active efforts to regenerate degraded forests.

During the early 1990s, forests in the North provided a total sink for 0.6–0.7 Pg of C per year (1 Pg = $10^{15}$ g = billion tons).

While forests in the North have the potential to increase carbon stocks, there are also areas of forest with none or little active forestry, where carbon sequestration may approach their maximum in the coming decades. Once that happens, forests may no longer be a net sink, but can for at least a period, become a net emitter, as mature and old forest naturally degrades and rejuvenates.

In the South, focus has been on a) and b) (see Table 4). Dominated by developing countries at different initial and middle stages of forest transition, demand for especially new croplands and pastures have driven deforestation, while harvesting of rich timbers, fuelwood and overgrazing have been the key factors driving net forest carbon losses. Mitigating these loss trends is what Reduced Emissions from Deforestation and Degradation (REDD) is trying to achieve. However, tropical forests that have not yet reached their growth climax will still accumulate carbon. This sink function could at early stages of climate change for certain forest types actually be enhanced, due to a combination of atmospheric $CO_2$ fertilization and marginally higher temperature (as long as hydrological balances are not compromised).

It has been estimated that global terrestrial ecosystems annually remove about 3 billion tons of anthropogenic carbon from the atmosphere, the bulk of which comes from forests, which could constitute about 30% of emissions from fossil fuel burning.

**Box 2. Carbon services through REDD: the *Bolsa Floresta* Program (Amazonas State, Brazil).**

Juma, Brazil´s first certified REDD project, started in 2007 as part of the Bolsa Floresta Program. Bolsa Floresta covers over 1 million hectares in 15 of Amazonas State's protected areas with human presence. The Juma Sustainable Development Reserve lies relatively close to a rapidly expanding agricultural frontier. Model-projected future deforestation is high for Juma, as cattle production is expected to gradually encroach onto its southern and eastern boundaries, so protecting the area yields corresponding carbon benefits from avoided deforestation and degradation. The Bolsa Floresta program engages primarily with the local population in the protected areas to promote good forest stewardship, through conditional conservation incentives and interventions aimed at improving quality of life. It combines integrated conservation projects with payments for environmental services (PES) and regulatory enforcement. Enhancing conservation alliances with local residents is supposed to also bolster the integrity of protected areas when pressure from outside increases, as the agricultural frontier gradually approaches. Evidence from older Amazon colonization frontiers suggests that stable forest-agriculture mosaics can emerge from smallholder-dominated landscapes, thus avoiding the more common conversion to extensive pasturelands. Bolsa Floresta is an attempt in that direction, and time will tell us to what extent it will succeed.

## Carbon storage enhancement in and outside the forest and fossil fuel substitution

Beyond the avoidance of deforestation, management of forests and their sustainable use can have important impacts on carbon balances. Under our heading "avoiding forest degradation", reduced impact logging is one vehicle for minimizing carbon emissions from timber harvesting. Under the heading of "forest enhancement" the furthering of e.g. continuous cover forestry systems relying on natural regeneration and more undisturbed undergrowth and soils generally increase average carbon stocks, but also forest management measures like enrichment planting in the regeneration phase.

To measure the full carbon benefits from forestry and forest management alternatives, also the end uses of forest products needs to be accounted for. Harvesting, transport and processing are of course energy consuming processes, and therefore have a negative carbon impact. However, construction wood and high-value furniture wood will remain in use and embedded in constructions for decades, sometimes centuries, and in that function also represent a lasting carbon storage. Thus, while extracting wood from the forests represents a reduction of carbon storage in the forest, it may imply an increase in storage outside the forest. At the same time, the construction wood may substitute other materials (steel, concrete), which in turn have sometimes quite significant carbon impacts in their respective production processes. The assessment of these production chains, storage and substitution effects is thus quite complicated.

Much wood, however, is harvested for shorter term uses, e.g. paper, is a bi-product from other wood-based production (sawdust, bark residues), or is directly produced for energy uses. Much of the wood from these sources go into the energy sector, either directly or after a couple of recirculation steps. In the energy sector, these forest biomass resources often replace a mixture of fossil fuels (mainly coal). If the forest waste products reaching the energy sector would alternatively have been left for natural decay or burning with no use of their energy content, the use of it for energy represent a carbon benefit. When growing short-rotation forest crops for energy purposes, there is also substitution in the energy sector, but again, the process of bringing forward the woody biomass to the energy plants is a carbon emitting process that needs to be integrated in the assessment, relative to the fossil fuels it may replace. Furthermore, from a carbon emission angle, the land may have been used for other purposes (e.g. long-rotation forestry) where net carbon effects could have been higher. Again the assessment of these production chain, storage and substitution effects is complex, and the conclusions will depend on the specific types of forest, and of alternative land and energy uses, among others.

It has often been argued in popular debates that wood represents a carbon-neutral source of energy, because forest regrowth eventually recaptures the carbon released from energy consumption. While this is true over time and aggregated over landscapes for some or even most forest management and wood utilisation combinations, it is not likely to be true for all, when alternatives are fully accounted for. For example, short term forest rotations systems for energy uses are likely to be a poor type of choice on land where long term forest rotations targeting end-uses with higher carbon substitution effects is possible. Thus, relative to the latter, the former choice has a negative carbon impact.

**Figure 5.** Forests store massive amounts of carbon, but the use of wood in many purposes also may reduce overall emissions. Balancing the carbon emission benefits from storage and use of wood is a key challenge in climate policy. Photos: CCat82, fotolia.com and uzkiland, fotolia.com

The consequences of these many conflicting impacts is that caution is needed when assessing forest carbon benefits from forest management changes and forest products uses.

Alternative forest management schemes need to be assessed in terms of their impact in $CO_2$-equivalents, including preferably both storage effects, forest product use effects, and possibly indirect land use effects. Yet, the technical challenges for accuracy in the assessment of net carbon impacts also increase greatly as we try to include all these different layers of impact.

Assessing storage impacts in the North is often done using stand-level models of volume growth in combination with expansion factors for roots, branches and other carbon storage compartments. Such models are often treatment sensitive allowing rather detailed assessments of storage development over time. In the larger forest areas of the South with a focus on avoiding deforestation and degradation, a somewhat more simple accounting point of departure is taken, where forest carbon stocks depend on two factors mainly: forest area size and carbon densities (see also volume 2).

The technology for assessing carbon stocks and their likely development in the immediate future are constantly improving as is the reliability of the estimates, and hence the estimated benefits. Similarly, our insight into carbon effects of possible substitution products (e.g. concrete, aluminium, steel in construction). The carbon stocks in the forest and the substitution effects in different industries (construction, furniture, energy) likely make up the major part of the carbon benefits from forest and forest products. Assessing the carbon stocks of forest products in its end-uses (furniture, houses etc.) is a very challenging issue currently receiving increased attention.

Finally, it should be noted that forest based measures for climate mitigation is only one among many policy options available. Carbon as an externality affected by forest management is investigated also in some of the NEWFOREX case studies, and will be addressed later in this volume.

### Key messages

- Forests play a crucial role in the global carbon flows, and their ability to store carbon as well as produce low emission products are of immense importance.
- Carbon impact as an ecosystem services has a clear unit of measurement – tons of $CO_2$.
- Striking the right balance between storing carbon in the forests, managing and using forests and wood intelligently is a key challenge.
- The full carbon effects of wood in use requires assessment of the different production chains and end-uses – system-wide – and must be compared to the storage alternative too.

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# 2.4

# Forests and recreational services

**Liisa Tyrväinen**

Forests are important environments for outdoor recreation across Europe and are considered one of the most attractive types of nature. Forests, within or near urban areas as well as in rural areas, provide aesthetic experiences and a pleasant environment for many outdoor activities. Experiences that are typically sought after are predominantly enjoying the natural scenery, peace and quietness as well as getting physical exercise. The resulting health benefits are increasingly important for urbanized societies where insufficient recovery from stress cause long-term health effects.

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Forest-based recreation and tourism are direct benefits to people, but also contributes to human health by reducing stress, and enhancing both psychological and physiological recovery.

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In urban and peri-urban areas forests contribute to the quality of housing and working environments and their benefits are reflected in property values. Indirectly, attractive natural landscapes and recreational opportunities of forests can promote tourism and enhance economic development in both rural and urban areas. Moreover, people pick berries and mushrooms, hunt and engage in many types of outdoor activities in forests. There are, however, large regional differences in the supply and accessibility of forests in and around European cities. Moreover, in land-use planning processes, the recreation benefits of nature areas are not fully acknowledged due to limited information about their value to the communities and regions and therefore, their provision is difficult to justify faced with competing land-use interests.

## Provision of recreation services

Recreation services can be provided mainly in two types of forests. Firstly, in forests where the main aim is to provide recreational services, but they are multipurpose in nature allowing also timber production. The managed recreation forests are often located in urban and peri-urban areas nearby the users. In many rural regions the nature-based tourism sector offers a growing number of job opportunities for local residents and diversifies the traditional rural livelihoods. In these areas maintaining or enhancing amenity values of forests may be a key objective to guide forest management decisions.

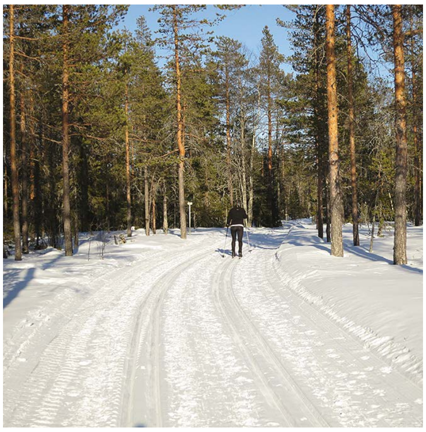

**Figure 6.** Forest-based recreation and tourism are direct benefits to people, but also contributes to human health by reducing stress, and enhancing both psychological and physiological recovery. Photos: Ruta Saulyte, fotolia.com and Linda Mantau.

> The landscape preference studies show that forest management, in particular intensive regeneration practices with clear-fellings, decrease the suitability of a landscape for recreation.

Landscape is a key attraction factor for forest-based recreation and tourism. In consequence, in recreation areas the quality of the landscape and the environment as such should meet the expectations of visitors. Therefore, some forest management is usually carried out in these forests, although less intensive than in commercial forests. In these forests hiking trails, signing and other types of services are also provided for visitors. The losses in timber production and investments in infrastructure are balanced against the higher numbers of recreational uses and improved recreation experiences perceived by the users.

Secondly recreation benefits are produced in protected areas, such as national parks, where the main aim is to preserve biodiversity and thus forestry operations are not allowed or restricted. National parks are popular tourism destinations with growing pressures to improve the recreational services for visitors. The provision and maintenance costs of an improved recreational infrastructure should be balanced against and exceeded by the values of increased recreational benefits also on protected areas.

> The nature of recreational services of forests varies between countries depending on the landowner structure and the distribution of use rights for many ecosystem services.

The possibility to use forests for recreation can be viewed as a public good, but not a pure one. In many countries the municipality or state may often be in charge of providing these services in designated forest areas. In principle, therefore, everyone has the

Table 5. Examples of how management changes in a recreation forest affect changes in amount and type of recreation (Δ denotes change).

| Change in characteristics | measure of change | observed recreation |
|---|---|---|
| Change in **infrastructure** | Δ amount of recreation routes and/or amount of recreation facilities | Δ increased number of visits<br>Δ length of visits |
| Change in **accessibility or legislation** | Δ amount of forests available for recreation | Δ number of visits<br>Δ number of user groups |
| Change in forest species | Δ change in quality of landscape and environment | Δ type of recreation activities<br>Δ types of user groups |
| Change in harvest regime | Δ less visible traces of forest management<br>Δ share of protected areas | Δ experienced quality of a forest visit<br>Δ increased number of visits<br>Δ types of users |

possibility to consume, for example, the pleasant wooded landscape, or has access to forest areas without paying an admission fee. In the Nordic countries, for example, some recreational services are public goods due to free access to all nature areas independent of the landownership. Thus there are no market values for these services. There is, however, a large share of forests in private ownership with limited public access for nearby users across European population centres and nature-based tourism destinations. There is a need to understand the demand and the value of the recreation services of forests to guarantee their adequate provision. For enhancing recreational uses and tourism based business development based on the use of land in private ownership, market-based mechanisms are needed. They are necessary to provide incentives and to compensate the landowners for the costs resulting from the recreational use of their forests or for undertaking landscape management measures enhancing the suitability of the area for recreation and tourism.

Information on user preferences for various recreation services or desired characteristics of the forest environment is needed for management and valuable for both society and the forest owners supplying the services. Research tells us that people prefer stands of tall and mature trees, but the preferred tree species relate to the specific region in question. In general, old and mature forest stands are preferred over young and small trees, but small trees forming the lower canopy layer of a two-storey stand are generally found to improve the aesthetic value of the stand. Variation in structure and species within the forests is greatly appreciated, as is also the combination of forests with fields, meadows and, in particular, watercourses at the landscape level.

## Quantification of recreation benefits

The basic data requirements for valuing recreation benefits include information like the number of visits, the accessibility of forests, the distances and means of transportation to the forests and their value in the recreational use. The most common approach to evaluate the recreation demand at the site level is visitor monitoring (visitor counting and surveys), where information linked to the use of a specific site is collected (Table 6). In Finland, for example, in each national park and hiking area a larger visitor survey is

Table 6. Examples of participation rates to forest recreation and average number of visits in Europe (Sievänen et al. 2008).

| Country | Rate of participation (%) | Annual average number of visits (visits/year) |
|---|---|---|
| Denmark | 91 | 38 |
| Finland | 90 | 120 |
| France | 72 | 15 |
| Germany | 66 | 37 |
| Norway | 76 | 44 |
| Switzerland | 96 | 76 |
| United Kingdom | 40 | 5 |

systematically collected every five years to monitor changes in the amount of visits and in their pattern. This survey includes information such as the length of the visits, type of activities as well as monetary expenditure related to the visit. The monetary expenditure is used to assess the local economic benefits of park visitations.

Recreation demand has also been analysed at regional or national level. National standardized surveys have, however, been conducted only in few countries mainly in the Nordic countries. For example, in Finland a national outdoor recreation demand inventory has been conducted in 1998–2000 and 2009–2010. Outdoor recreation statistics provide information of the recreation demand and its changes e.g. participation in 86 different outdoor activities, outdoor recreation nearby home and nature-based tourism. 96 percent of Finns participate in outdoor recreation, on average 2–3 times per week, summing up to around 170 times per year. Walking, swimming, spending time at vacation home or shore, cycling, berry picking, and skiing are among the most popular outdoor activities. In other countries, e.g. Denmark, similar national surveys have taken place several times.

Furthermore, the recreational quality of a site is an important factor affecting the amount and the types of recreation. Length of trails, the availability of camp grounds and other recreation facilities explain the number of visits to national parks or recreation areas. Moreover, the naturalness or management intensity affects the recreational attractiveness of a forest. In urban and peri-urban forests, appreciated characteristics of forests often link to accessibility, safety, tidiness and active management of the area, whereas in protected areas the requirements concerning biodiversity, richness of flora and fauna and aesthetic environments are expected to be more important.

In quantifying recreation benefits of forests there is a need to understand, how the changes in different quality attributes of managed forest areas affect the frequency of visits to the forests, and how citizens value the different characteristics of the managed forest recreation areas such as the presence of dead or decaying wood and trees, the share of deciduous trees or the share of protected and unmanaged areas. Moreover, data about how often citizens visit different types of recreational forests is needed, the existing potential substitutes around forests in question etc. In monitoring recreation benefits in Europe standardized methods and development of recreation benefit indicators are needed.

## Key messages

- Recreation opportunities are among the key forest benefits enjoyed by the general public.
- The provision of recreational benefits can be improved by increasing the accessibility of areas, improving infrastructure such as the availability and quality of of trails, campgrounds and other facilities.
- The quality of the forest environment for recreation can be improved through small-scale management practices.
- Quantification of recreation benefits is based on information on number and length of visits, type of activities and valuation of each visit by recreationists or tourists.
- European standards for assessing recreational values are called for.

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# 2.5

# Forests and biodiversity externalities

**Jette Bredahl Jacobsen, Anna Bartczak and Marek Giergiczny**

Biodiversity is the diversity within living organisms, and we typically distinguish between this diversity at different levels – from genetic diversity, to species diversity, habitat diversity, or even diversity at a landscape level. If we want to talk about how to quantify biodiversity, we first need to define the level of biodiversity we look at: Is it, e.g. the number of species in a forest area, or the number of different habitats available? Biodiversity enters into all four categories of ecosystem services (see Section 2.1), e.g. it is *supporting* decomposition, *providing* game meat, it is *regulating* water flows and it has a key function as a *cultural service* in terms of providing a basis for existence values and recreational uses. As opposed to ecosystem services, such as recreation and water, the values related to biodiversity protection are not only linked to active use. Rather it is often seen as a prerequisite for the provision or existence of other services, or in terms of supporting or regulating services. Therefore, when quantifying the ecosystem service "biodiversity" or "biodiversity protection" it is important to keep in mind which aggregation level we target and what the actual services are, we expect to value. This is needed to avoid double counting and to ensure consistency between measurements of biodiversity and the values assessed. Many services linked to biodiversity are essential for the functioning of any ecosystem. It is for example difficult to imagine a stable forest ecosystem without decomposition.

From a forest management perspective we are often interested in evaluating the outcomes of changes in management practices, when evaluating if better provision of ecosystem services are worthwhile. Therefore it is crucial to be able to quantify marginal changes in biodiversity measures as a result of management changes. The provision services relying on aggregated biodiversity are the most easily quantifiable e.g. cubic metres harvested, kilograms of berried collected, etc. and will not be addressed further here. The quantification of regulating and supporting services often requires specific biophysical measures, as described e.g. in the above chapters on water regulation and carbon. It can be difficult to distinguish one service from other services, and double counting is always a real risk.

Cultural services apart from recreational activities are probably the most intangible and therefore difficult to quantify. The use-component of biodiversity as a cultural service consists of the joy people have when e.g. seeing a beautiful view from the motorway, or visiting divers landscapes, where biodiversity is a crucial component for the quality of the experience. The non-use part of biodiversity protection consists of the value people attach to nature, e.g. based on its role in the culture. For example, many countries have a strong picture of nature as an inherent part of their national history supported

Table 7. Examples of how changes in forest characteristics affect changes in ecosystem functions and services and how these may be measured (Δ denotes change).

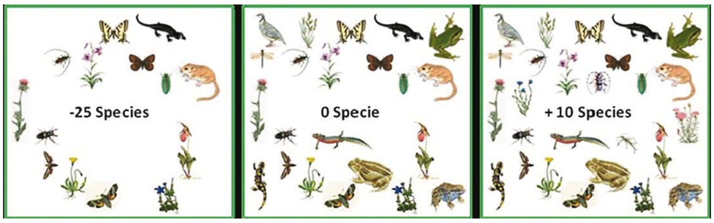
| Change in characteristics | ... Ecosystem Services | Measure of service |
|---|---|---|
| Change amount of dead wood in the forest | Δ conservation of endangered species (existence value)<br>Δ Natural cycles like decomposition (functional and existence value)<br>Δ Aesthetic values | Number of species conserved<br>Decomposition rates or Quality index for natural cycles<br>Structural index or visualization of a forest |
| Change in species composition (more native species) | Δ conservation of endangered species (existence value)<br>Δ Natural cycles (functional and existence value)<br>Δ Aesthetic values<br>Δ Conservation of genetic diversity | Number of species conserved<br>Decomposition rates or Quality index for natural cycles<br>Structure of a forest<br>Indices for genetic variation |
| Change in population levels of given species | Δ Genetic diversity – option & existence values<br>Δ Recreational value – increase the chance of encountering<br>Δ Species conservation (existence value) | Number of animals or percentage change<br>Number of animals or percentage change or a chance of encountering or Aesthetic values<br>Number of animals or percentage change |
| Change in amount of open grazing areas in the forest landscape | Δ Change in aesthetic view<br>Δ Change in species diversity at landscape level | Length of inner and outer forest edges<br>Number species present at forest landscape level |

by its presence in art and literature. More importantly perhaps, among the values associated with biodiversity protection is the value we as humans assign to the mere existence of species or habitats – e.g. the joy we get from knowing that a certain species exist, even if we are never likely to see it.

Often, changes in the level of biodiversity are not broken down into its functions, but are measured as physical changes in ecosystems, for example the area covered by a specific forest habitat, the number of species preserved, or the proportion of trees left for natural decay. One reason for this is that often a given change affects several of the ecosystem services relying on biodiversity. Table 7 gives some examples of how given changes affect the ecosystem services and how it has been quantified in the literature. Table 7 clearly shows that that a management change will often affect several services.

It is a challenge to find a good measure of changes in the quantity of biodiversity – which encompasses the various values. Sometimes, the relevant measure is absolute, for example conservation of one species, but at other times it makes more sense to talk about relative measures, e.g. a percentage change in area use. Sometimes more complex measures are used such as e.g. the Shannon forest age diversity index or the Shannon species diversity index. Those indices of diversity take into account richness, evenness of species distribution and trees' age classes. The higher the index, the richer and more evenly distributed is the age and species classes. The extent that such measures are useful for quantifying the changes in ecosystem services provided, depends on what kind of

Table 8. Measures used to quantify the ecosystems service biodiversity in NEWFOREX.

| Biodiversity attribute | Measurement of changes | Graphical illustration |
|---|---|---|
| Number of species | Changes in species' number (in absolute numbers or in %) |    |
| Naturalness of forest process | Descriptive levels (low, medium, high) |  |

values dominate the ecosystem services. Measures such as the species indices are relatively rarely applied in valuation surveys as they are complex concepts and ordinary people can encounter difficulties to conceptualise and relate to them.

Naturalness or natural processes is another way of describing the functional services of biodiversity. Again a quantification measure is not obvious, but often descriptive levels are chosen for this kind if the aim is to use it for valuing a bundle of ecosystem services associated with the degree of naturalness. Often illustrations (such as pictograms) or photos are used to help in communication of the different levels of biodiversity changes. Especially, the visual communication applies to studies focusing on landscape diversity. Table 8 shows some examples of pictograms associated with biodiversity quantification, which were used in the NEWFOREX project.

One final aspect to consider when quantifying the ecosystem services arising from biodiversity protection is when the service appears. It may take several decades before a given change in land use management actually brings about the desired ecosystem services to be provided. The measures of biodiversity – like secured survival of an endangered species – will typically occur much later than the management initiative is taken. That also means that there inherently is uncertainty associated with the measures – will the species actually survive with this measure? Or will other factors cause that they do not survive, regardless of the management change being implemented? Because of that the measures are often difficult to verify in the short run.

## Key messages

- Forests are important habitats for much of Europe's biodiversity and forest management is crucial for its preservation.
- Many benefits from our forests rely on the existing biodiversity in complex ways, and the citizens of Europe care about the protection of diversity as such.
- The number of endangered species or the area of an endangered habitat protected is often used as a measure for biodiversity services resulting from management measures.
- The degree of naturalness and natural processes in a forest area is another option sometimes used to capture the bundle of services and functions related to management measures.
- The best way to quantify biodiversity impacts depends on the kind of values targeted.
- Improvements in biodiversity as a result of management often appear only after a long time horizon.

## Recommended reading

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# 3.

# Valuation of ecosystem services

## 3.1 Why should we estimate the value of ecosystem services?

**Robert Mavsar and Elsa Varela**

Environmental valuation attempts to assign monetary value to the whole range of ecosystem services, including those that have no established market values (e.g., biodiversity protection, watershed protection, aesthetics, and recreational uses) and are not considered in traditional economic valuation frameworks.

The Total Economic Value concept should guarantee that all ecosystem benefits are considered.

However, the wide range of benefits forests provide creates challenges for the analysis. Thus, a coherent analytical framework, based on the concept of Total Economic Value (TEV), has been developed to ensure that benefits can be considered systematically and comprehensively, while avoiding double counting. In recent years, the TEV has been widely used as a framework for trying to quantify the full value of the different ecosystem components and processes. In general, the TEV framework disaggregates the values into use and non-use values. Use values are related to the direct, indirect or future (option) use of a natural resource. On the other hand, non-use values, also referred to as "passive use" values, are values that are not associated to the actual use or even the option to use a good or service, and do not require the individual person to undertake specific actions or carry specific cost to enjoy them. These values are derived from the knowledge that the natural resource is preserved.

An important fraction of ecosystem services are traded on markets and their value is defined by the market price. The majority of ecosystem services are, however, supplied to the society or various groups of users, for free or at a price which is significantly lower

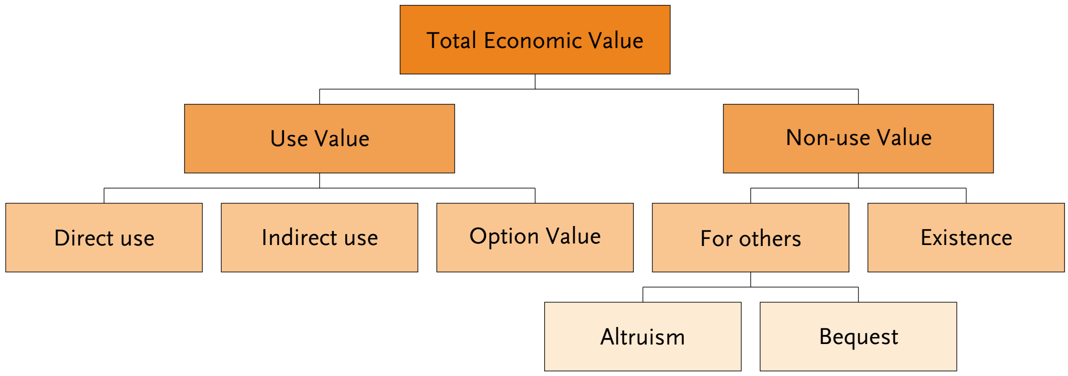



**Figure 7.** Total economic value framework.

than the costs associated with their provision. In such cases (no market price) "alternative" valuation approaches are applied to estimate the economic value. In one of the following chapters we briefly outline the principles of these methods.

Although the economic valuation can be seen as a useful tool in many situations, there are some significant limitations, which should be considered when conducting economic valuation or using the estimated values.

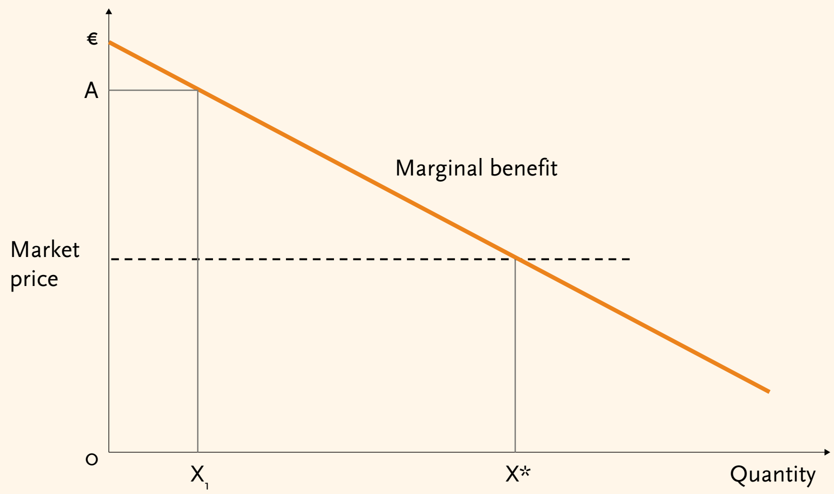
The economic valuation is only valid for small changes in the provision of ecosystem goods.

The appropriate context for economic valuation is conditioned, among other things, by the scale of environmental changes. Monetary valuation is most meaningful when considering small, or marginal, changes in the provision of ecosystem services or ecosystem characteristics. For example, determining the value of a preserved forest at a local scale is more reliable and helpful than attempts to determine the global value of all forests. The loss or degradation of forests on a local or regional scale is imaginable, and the consequent loss of services may not result in such dramatic alterations in ecosystem processes as to place human survival at risk. In contrast, the loss of all forests on a global scale would result profound consequences, which are beyond the scope of the analysis and beyond the range where monetary valuation methods can be validly applied. Therefore the question as to what is the 'value of everything' makes little economic sense.

Another important issue of economic valuation is that the resulting estimates are often highly subjective, being sensitive to both the methods selected and assumptions used. For example, some valuation methods mainly focus on marketed goods and services, but omit non-market values. In addition, the selected ecosystem services, valuation period (number of years) and discount rate (how we value the future) have profound effects on the estimates.

### Box 3. Economic value vs. market price.

It is often erroneously assumed that market price measures the economic value of a good or service. However, the market price only tells us the minimum amount that people who buy the good or service are willing to pay for it. When people purchase a good, they compare the amount they would be willing to pay for that good with its market price. They will only purchase the good if their willingness to pay is equal to or greater than the price. Many people are actually willing to pay more than the market price for a good, and thus the value they place upon a good exceed the market price. The following figure shows the difference between the market price and value, where the red line indicates the marginal benefit people obtain from a good and the dotted line the market price of a good. Furthermore, if a consumer values the unit X1 of the good at A, s/he would still only have to pay the market price, retaining the difference between A and market price for their own pleasure.




The same assumption would hold for each unit of the good up to the point where the marginal benefit curve and the market price are equal (unit X*). After that point the consumers' value for the good is lower than the market price. Thus, the consumers would not be willing to pay the requested amount.

In addition, inaccuracies exist because of incomplete understanding of complex ecosystem processes and inherent biological uncertainties (for example, how much forest is required to provide sufficient flood regulation or water filtration).

The economic valuation can be used for various purposes, but not for establishing the price of an ecosystem service.

Thus, before anyone decides whether or not, how to conduct an economic valuation study for a specific policy case, it is important to have a clear idea about the main aims of such an exercise. There are a number of situations where economic valuation can be useful for policy makers and others. The most common objectives are:

1. *Awareness rising about the contribution of ecosystems to the social wellbeing.* Although in past decades there were major improvements in the way people perceive the benefits that ecosystems provide, still significant improvements can be achieved. In particular when developing policies or management measures, and spending public funds, it is important to obtain a wide support from the general public. Valuation studies can assess and communicate that.
2. To obtain *information about the relative importance of ecosystem services* and *preferences for their provision across and from different stakeholder groups.* In particular in environmental management planning this is an essential issue. Typically, when deciding about management alternatives at least two basic inputs need to be considered. On one hand, the needs and preferences of different stakeholders are the base for defining management objectives. While, on the other hand the characteristics of the natural resource (e.g. a forest area) helps to define what is feasible in terms of ecosystem provision.
3. *As a decision support tool for assessing the relative economic impact of alternative actions/policies.* The latter can provide a way to justify and set priorities for programs, policies, or actions that protect or restore ecosystems and their services. This type of valuation can provide useful information to policy-makers by highlighting the economic consequences of an alternative course of action.
4. *Identify potential winners and losers* when adopting a certain management alternative. Decisions related to the management of natural resources commonly affect a number of stakeholders. In general we are always striving that the overall effect of such decisions is positive (e.g. increasing the overall social welfare). Nevertheless, part of the society can suffer adverse effects and a decrease of the wellbeing. Thus, it is important to evaluate how the proposed decisions will influence the society's income or wealth distribution. A negative impact could be that the proposed measure/policy would decrease the income/wellbeing of the less wealthy part of the population and thus contribute to a more unequal society. For example, the protection of a natural area can contribute to the improvement of water quality for downstream (urban) users. However, at the same time it might limit the income generation opportunities for the rural population.
5. *Evaluating the impacts of environmental policies.* This could include evaluating the ecosystem service costs associated with habitat conversion, runoff, or pollutant discharge. It could also include looking at the benefits of increased investment in enforcing environmental regulation and in strengthening resource management. For example, Natura 2000 is an EU wide network of nature protection areas. The aim of the network is to assure the long-term survival of Europe's most valuable and threatened species and habitats. In this context, the EU Rural Development Policy foresees compensation payments to land owners of Natura 2000 sites. In the past six years (2007–2013) the European Union spent about 64.9 million € on this measure. However, a study estimated that the overall benefit of different ecosystems services (climate change mitigation and adaptation, improvement of water quality, food provision, job creation and livelihood, health and social cohesion) is in the range of 220 to 310 billion € per year.
6. *Establishing incentive schemes or markets for ecosystem services.* A significant part of ecosystem services has the characteristics of public goods. This means that it is impossible or very difficult to prevent anyone to use them. The free accessibility of ecosystem services implies that the provider (e.g. land owner) is not com-

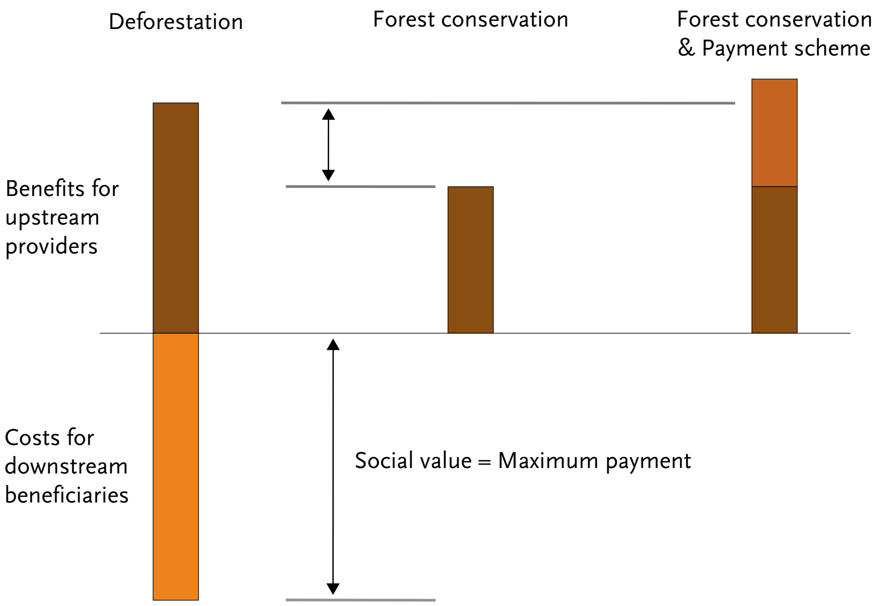



**Figure 8.** Illustration of the payment range for a compensation scheme (modified from Pagiola et al. 2004).

> pensated for their provision, and is thus not motivated to manage the forest in a way that would optimise the quantity of these non-market goods and service. To correct this situation, compensation payments or other incentives can be applied. However, before such incentives can be established, an understanding and estimation of the social benefits is necessary. However the estimate of an economic value should not be considered as the price (compensation payments) of the ecosystem service. The estimated value merely indicate the social value and thus the maximum amount the society would be willing to pay to guarantee the provision of an additional unit of the valued ecosystem good.

Figure 8 shows a simplified example of how the amount of compensation payments should be established. The figure presents three different scenarios. On the left hand side we have the "Deforestation scenario" where a land owner would harvest all the wood from his forest (clear cut). This would yield him benefits as shown by the green bar. At the same time, the clear-cutting would decrease the watershed services provided by this forest, and consequently the quality of the water for downstream users. The economic value of this loss is indicated by the red bar. The "Forest conservation scenario" foresees that the land owner would cut only a part of the trees (e.g. selective cutting). In this way, he would maintain the provision of the watershed services as before harvesting, but he would lose part of the income from wood production. Thus, the land owner would lose part of his income, but there would be no benefits lost for the downstream users. Finally, the third scenario "Forest conservation & Payment scheme" the harvesting intensity would be the same as in the second scenario, but the land owner would also receive a compensation for the lost income. An economically efficient compensation payment should be higher than the land owner's loss of income, but lower than the estimated social value of the provided watershed services. The actual amount would be negotiated between the providers and the beneficiaries.

## Key messages

- The economic valuation aims at assessing the monetary values of ecosystem services.
- The multitude of provided ecosystem services requires a coherent framework, and the Total Economic Value frame enables us to consider all different types of values systematically.
- Although the economic valuation is a very useful tool in the decision making process, it is only valid under certain circumstances.
- The economic valuation should be only applied to value smaller changes in the provision of ecosystem services, which however still leaves a wide range of cases where economic valuation can be applied.
- Environmental valuation can serve to raise general awareness about the importance of ecosystems for the public welfare, to help to identify public's preferences for the provision of different ecosystem services, and to inform the design of policy instrument targeting ecosystem service provision.

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# 3.2

# Methods for assessing the values of ecosystem services

**Jette Bredahl Jacobsen**

Valuation of ecosystem services relies on an anthropocentric approach, where an ecosystem has an economic value when it provides services to humans. Assessing this value relies on a proper quantification of the ecosystem services as discussed in section 2. Without proper quantification, valuation makes little sense. As for quantification, we would often look at valuing the effect of policies rather than the total value of a given service. Many ecosystem services are not marketed and hence we need to use methods that are able to capture this. Ecosystem services are of economic value to society when they in a very broad sense provide utility to humans. To assess their value we should therefore look at people's preferences for various levels of ecosystem services. An alternative to this preference based approach is to use a cost-based approach, though it per definition measures the costs of providing an ecosystem service rather than its value. We have three main approaches to value ecosystem services – the opportunity cost approach, and two preference-based approaches, revealed preference methods and stated preference methods. A search on Web of Science using the keywords "environmental valuation of ecosystem services" terms showed 205 studies using opportunity cost approaches 140 revealed preference studies, and 360 stated preference studies.

The opportunity cost approach measures the cost of the alternative provision method for a good or service. For example consider the value of filtration services of drinking water from forests; the alternative opportunity cost to securing this service could be the cost of cleaning the water at a water treatment plant instead prior to supplying it to consumers. Another example could be the assessment of the value of pollination. In this case, the opportunity cost could be the production loss if it was not present. The opportunity cost approach is widely accepted and easy to understand, and typically based on actually monetary losses. However, it only considers the cost side, not the benefit side, and usually only considers those costs that are measureable in markets. Therefore estimates from this method are not directly comparable with other valuation methods and cannot serve as a direct input to a cost benefit analysis.

Many ecosystem services have a value because for many different individuals in the society – the individual's value of a walk in the forest; the individual's value of preserving a species or seeing a beautiful landscape. The value for society of all these values therefore consists of the sum of the values across all individuals. To measure this we have the revealed and the stated preference methods, see Figure 9.

The revealed preference methods elicit the use values of an ecosystem typically by measuring the costs individuals are willing to care for enjoying its services. There are two main approaches (see Figure 9), the travel cost method and the hedonic price method.

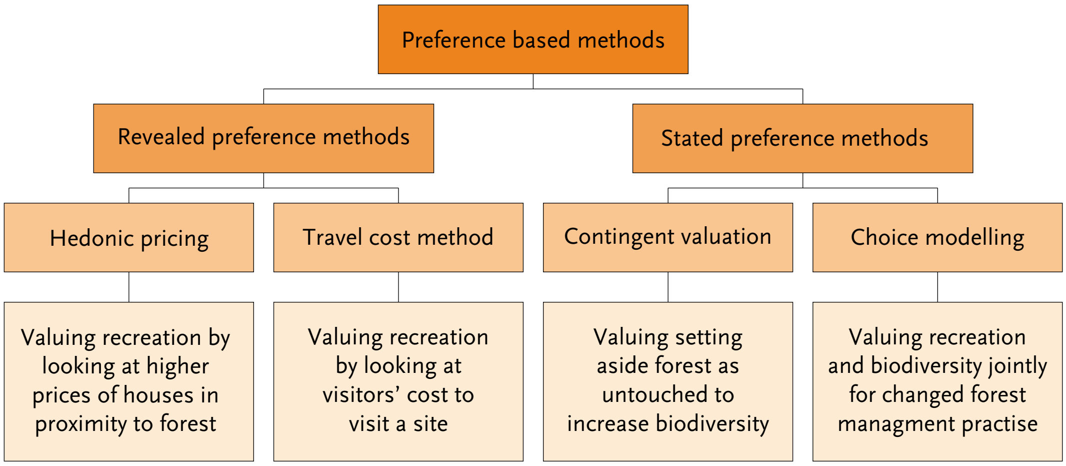


**Figure 9.** Different methods to assess the value of ecosystem services by looking at preference-based methods. The lowest level indicates examples of the different methods.

The travel cost method makes it possible to derive the recreational value of a given site from the travel costs incurred for the individuals visiting the area. The advantage of this method is that it relies on peoples' actual behaviour, and it is therefore unquestionable that the recreation does have a certain value if people are willing to travel for it. The challenge therefore lies in estimating the right cost or extra price given that many activities have multiple purposes and each site often have multiple alternative sites. For example the value of a Chinese visiting the Finnish forest is likely not equal to the price of the airfair from China to Finland, as such a long trip would typically involve other activities as well as costs. One can also question whether to consider actual travel cost or people's perceived travel cost. Quite many people would e.g. not count depreciation of a car as a cost of travelling, even if it is. Thus, the value people assign to their use of recreational sites is likely better reflected in their perceived costs than in formal approximations of their costs. Yet the former is harder to obtain. Like the travel cost method, the hedonic pricing method relies on actual behaviour. It can for example be used for estimating the value of proximity to a forest by inferring it from how much more houses in proximity to forests cost, compared to houses further away. Also here it becomes a challenge to isolate the effect of the ecosystem service in question, and advanced spatial statistics and detailed spatial data are required.

The main limitation of both revealed preference methods is that they can only estimate actual use values, e.g. values that you csn only enjoy by buying a house with these environmental attributes, or by making that trip to the forest for recreation. Thus we only get the value from those people who actually use it. That means that if we attempt to assign environmental values to a forest using only house price data, we only consider the values enjoyed by people very close by – typically within less than 6–700 m from the forest. This is a source of underestimation and bias as people further away may also experience a (albeit most likely somewhat smaller) value from that forest. Likewise, if a person enjoys the option value that he could go to the forest for recreation even if he doesn't actually do it – we would not easily include his value as he is not likely to be

Which initiative do you prefer?

| | Existing forest management continues | Initiative 1 | Initiative 2 |
|---|---|---|---|
| Access on foot outside road and path | On 25% of the area | On 25% of the area | On 100% of the area |
| Amount of the 660 endangered species that are secured survival | 0 species secured survival | 0 species secured survival | 100 species secured survival |
| Possibility of natural processes in the forest | Low level | Low level | Medium level: 5 trees are left to natural decay per ha |
| Increased production of groundwater, measured in number of house-holds' consumption | Same amount of groundwater as today | Same amount of groundwater as today | Same amount of groundwater as today |
| Extra income tax per year for your household | 0 DKK | 250 DKK | 1250 DKK |
| Choose one | ☐ | ☐ | ☐ |

**Figure 10.** Example of a choice set in a choice experiment in the Danish case study of NEWFOREX. Levels of the attributes vary from choice set to choice set. Each respondent typically receive 6–12 choice sets.

present in the samples of e.g. travel cost studies. Finally, and most likely more importantly, it is a potential problem that pure non-use values cannot be estimated using revealed preference methods. Thus it is not possible to value for example the joy one has from knowing that the existence of a species or a habitat is secured, or from knowing that other people and e.g. future generations may derive a pleasure from experiencing them.

That only use values are measured also implies that it is not possible to directly measure the recreational values of a nature area that is being established *before* it is actually established. We have to resort to assess it using benefit transfer methods if such are available with reasonable reliability. This is an important limitation as we are often interested in exactly knowing the value to use it for deciding decided whether to establish it or not.

Stated preference methods are particularly useful for valuing ecosystem services from forests as these contain many non-use elements. For example, the value of biodiversity protection or the value of changes in future recreational options and bequest values. The stated preference methods essentially ask people, in a questionnaire or interview format, to perform hypothetical trade-offs between policies, varying in costs and sometimes also characteristics. This allows the analyst to estimate willingness-to-pay for the ecosystem services. Two main methods of stated preference methods exists; contingent valuation and choice modelling (choice experiment). For the contingent valuation method, respondents are asked to place a bid or accept a bid for a described policy. In the choice modelling approach respondents are asked to choose among policy alternatives with varying levels of ecosystem services and costs associated. An example from the Danish case study in NEWFOREX is shown in Figure 10. As is seen, respondents are asked to trade-off ecosystem services against a price, but also trade-offs across ecosystem services.

The main disadvantage with the stated preference methods is exactly that it is stated, and thereby hypothetical. For many people it is difficult to say "no" to a good cause, especially when it is costless to do so. The methods are being developed so as to minimize this so-called hypothetical bias, but it remains an issue to be aware of. Furthermore, it is often difficult to get respondents to relate to the quantity of the ecosystem service being provided. This sensitivity to scope is therefore often a main issue to address in valuation surveys to ensure validity.

Performing valuation surveys is costly. Therefore benefit transfer is often being used in decision making, where the estimated value of ecosystem services in one location is transferred to another. The studies conducted in NEWFOREX may also serve that purpose. Databases exists that collect results from valuation studies around the world. The probably most well-known of these is evri.ca.

## Key messages

- Several environmental valuation techniques are available, but careful tailoring for the specific forest ecosystem services in focus is always needed.
- Preference based approaches estimate the value of ecosystem services based on observable data about peoples' behaviour and choices in relation to situations where they may enjoy the services.
- Revealed preferences are useful for estimating values of ecosystem services relying on peoples active use, but cannot estimate non-use values arising from e.g. habitat or biodiversity protection or concerns for future generations.
- Stated preference methods are able to captue both use and non-use values resulting from ecosystem services.
- Because stated preference methods are hypothetical, special caution has to be given in the design of valuation effort to avoid possible biases.

## Recommended reading

Bateman, I.J., Carson, R.T., Day, B., Hanemann, M., Hanley, N., Hett, T., Jones-Lee, M., Loomes, G., Mourato, S., Özdemiroglu, E., Pearce, D.W., Sugden, R. and Swanson, J. 2002. Economic valuation with stated preference techniques. Edward Elgar, Cheltenham,UK. 458 p.

Hanley, N. and Barbier, E.B. 2009. Pricing nature, cost-benefit analysis and environmental policy. Edward Elgar, Cheltenham, UK.

Riera, P., Signorello, G., Thiene, M., Mahieu, P-A, Navrud, S., Kaval, P., Rulleau, B., Mavsar, R., Madureira, L., Meyerhoff, J., Elsasser, P., Notaro, S., De Salvo, M., Giergiczny, M. and Dragoi, S. 2012. Non-market valuation of forest goods and services: Good practice guidelines. Journal of Forest Economics 18: 259–270.

The Swedish Environmental Protection Agency. 2006. An instrument for assessing the quality of environmental valuation studies. 120 p.

# 3.3

# Valuing water externalities from forests

**Bo Jellesmark Thorsen**

There is a complex interplay between forests and water cycles and there are many different forms that water externalities may take. This complicates the task of measuring and assigning values to the changes in water externalities that may result from changes in forest management. The natural science aspects of interactions between forest management and larger water cycle dynamics and externalities are only partly understood and in many cases certainly not quantified, whether in terms of general models or even case study estimates. The different forms that water externalities may take also makes various approaches to valuation relevant and the choice of valuation method should be adequately matched to the kind of services in play to capture values correctly.

In a recent review Ojea et al. (2012) categorizes and analyses the work undertaken in 36 studies of forest water externalities. A dominant part of these studies address provisioning services mainly in the form of water availability (quantity, quality and stability of supply) for households and occasionally industries for their consumption, but also supply (stability) of water for hydropower uses, as well as the absence or reduction of sedimentation reducing provisioning services. A large part, however, also considers regulating or supporting services, mainly related to the regulation of water levels in streams and wetland habitats. Finally, a few also include, or simply focus on, cultural services in the form of mainly recreational values dependent on forest-water dynamics.

It is important to distinguish carefully between these two central concepts: the value of water to society and the price or cost of water provision.

It is common place to convert value estimates to values per forest hectare (and year in many cases). There are two reasons: firstly, it may in many cases make valuation numbers more comparable on a spatial scale. Secondly, it enables a more direct comparison of values with the costs of the forest management actions considered, which are very often assessed on a per hectare basis. A note of caution, however, would be advisable here, as it may lead to the general assumption that the externality provided is linear in the area treated. For some water externalities and for homogenous forest areas this may be a reasonable approximation. There are other cases, however, where such a direct link between the scope and value of the water externality provided and the size of the forest area is not an adequate assumption, e.g. for the provision of groundwater at landscape level or scale dependent regulating functions for habitat protection.

Figure 11. Forest ecosystems impacts watershed dynamics through run-off and groundwater recharge. The resulting impacts on water related services can have great value. Photo: Katerina Zerva.

In Table 9 we show examples of approaches applied in earlier studies and one of the NEWFOREX case studies from 2013. The table illustrates the range of methods and approaches applied in different case studies. When the focus is on ecosystem services that benefits e.g. water provision companies or hydropower companies, the approaches used may adequately address the opportunity cost of these companies in terms of the costs of compensatory measures like purification, sediment removal, stream controls or similar. Alternatively they may asses the income foregone in terms of the potential improved income from better water capacity management. In cases where focus is on the enhanced provision of water for household consumption, we often find studies applying stated preference methods like the contingent valuation method or choice experiments. Another feature also evident from Table 9 and the existing literature is that while the literature contains numerous case studies of forest ecosystems' impact on water externalities in developing countries, there are much fewer studies of this aspect in developed countries.

## Case illustration: Valuing forest groundwater provision in Denmark

Forests and forest management systems play an important role in the watershed management in Denmark in ways that have been investigated thoroughly as the country relies almost exclusively on clean groundwater for most consumptive uses. For example, afforestation measures are often used as a land-use change driver to switch agricultural

Table 9. Examples of studies valuing various types of forest water externalities, using various valuation methods.

| Authors | Country | Ecosystem services addressed | Measure of service | Valuation method |
|---|---|---|---|---|
| Adger et al (1995) | Mexico | Reduced sedimentation and erosion in water ways | Tons of sediments pr. hectare of forest | Avoided cost of water purification |
| Campbell et al (2013) | Denmark | Groundwater (for consumption) recharge from forest areas | Quantity of recharge for case area ($m^3$/ha) | Choice experiment addressing households |
| Chomitz et al (1999) | Costa Rica | Stabilization and regulation of water flows for hydropower | Hectares forested | Companies' WTP for forest cover |
| Johnson and Baltodano (2004) | Nicaragua | Stability, quality and quantity of potable water for households | Improvement – but not quantified | Contingent valuation addressing households |
| Lundhede et al (2010) | Denmark | Forests ability to reduce nutrient leaching to groundwater | I/O of purification of groundwater | Choice experiment addressing households |

land into permanent forest reserve lands, when such lands are situated on top of important peri-urban groundwater reservoirs. While the result is in general to reduce the speed of groundwater recharge (at least in the long run), the reduction in the risk of nutrient and pesticide contamination of the entire groundwater reservoirs is considered much more important, and in addition such peri-urban forests have other benefits like recreational and amenity values.

However, also inside existing forest lands, forest management measures can affect groundwater measures of relevance. In much of the case study area of the Atlantic case study in Denmark, groundwater for drinking water is becoming increasingly scarce, partly due to increasing demands from the urban populations and partly due to increasing number of wells being contaminated on agricultural lands. Thus, enhanced groundwater recharge to fill reservoirs from the top faster will be considered a benefit. Recent studies indicate that switching tree species from conifers to broadleaves will enhance annual groundwater recharge with as much as 200 mm/year, due to the lover evapotranspiration of broadleaves. This will amount to as much as 2,000 $m^3$/year and hectare of forest switching tree species. On this basis a credible set of possible forest policies can be described that include the provision of enhanced groundwater recharge for drinking water in the case area.

Based on current tree species distributions in the Danish case area, it was assessed that policy changes could likely result in an additional groundwater recharge of up to 40,000,000 $m^3$ annually, corresponding to roughly the direct consumption of some 400,000 households. Therefore, a valuation study applying a Choice Experiment elicited the willingness to pay (WTP) for additional annual groundwater recharges amounting to 20,000,000 $m^3$ or 40,000,000 $m^3$. The WTP was elicited from a sample of people representative to the Danish population and assessed at the household level.

The Danish households are accustomed to pay for water consumption and typical payments amount to around 7€/$m^3$. This payment, however, mainly reflects the costs of providing the water, water piping, cleaning as well as costs related to getting rid of the waste water, like piping of waste water, storage and treatment facilities. Furthermore,

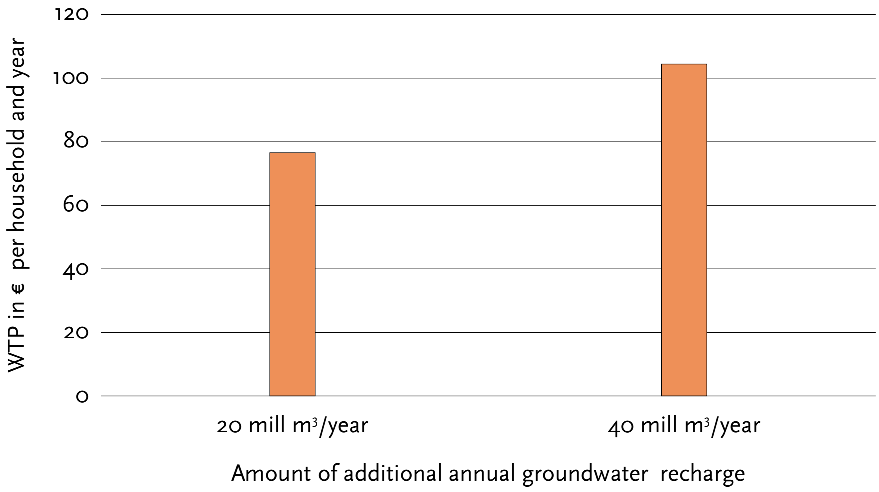


**Figure 12.** The estimated mean willingness to pay – WTP (in euro/year and household) for additional groundwater recharges of 2 × 106 $m^3$/year and 4 × 106 $m^3$/year. The mean WTP for 2 × 106 $m^3$/year is 76.5€/year and household and for 4 × 106 $m^3$/year, the mean WTP is 104.4€. Adapted from Campbell et al. (2013) using a full preference heterogeneity model.

the costs include various taxes and fees to cover e.g. measures to protect groundwater reservoirs. There is no resource rent as such in the water prices, in spite of the water being semi-finite in supply within given periods and also in spite of externalities related to the impacts on water levels in streams and lakes, often being lowered considerable during dry seasons of high consumption.

In Figure 12, we show the mean WTP of the Danish population for additional clean groundwater under the forests of the case area.

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The mean WTP for additional annual groundwater recharges amounting to 20,000,000 $m^3$ or 40,000,000 $m^3$ is estimated to 76.5€ and 104.4€ / year and household, respectively.

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The difference in mean WTP is statistically significant. However, there is considerable preference heterogeneity in the population around these values. Based on these estimates, some further calculations can set the obtained values into perspective. The number of households represented by the sample is approximately 2.3 million. Multiplying this with the obtained WTP estimates and dividing by the annual groundwater recharge quantities, we find that the estimated willingness to pay measures corresponds roughly to a willingness to pay of 6–8.8€/$m^3$ of additional groundwater. This is not far from the costs that respondents have experienced paying for water consumption in the household.

## Key messages

- Forests and forest management influence watersheds, circulation and water related ecosystem services in numerous way.
- Yet, the value of the water related ecosystem service depends strongly on the context and role of the water.
- In areas where groundwater is crucial for e.g. drinking water, changes in forest management may ensure groundwater improvements with values for society that rival the value of all other forest products.
- In areas, where flood and erosion control are crucial, the role of forest management may be to reduce the negative impacts of land uses on society.

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# 3.4

# The value of carbon sequestration

**Robert Mavsar, Elsa Varela, Davide Pettenella, Suzanne Elizabeth Vedel and Jette Bredahl Jacobsen**

Carbon sequestration is a global public good, as no one can be excluded from enjoying the benefits of climate change mitigation and no rivalry exist either. This means, that one tonne of carbon sequestrated in Sweden has the same value to any individual as one tonne sequestrated in Spain or Brazil. To estimate the economic value of carbon sequestration there are three main approaches, (i) the social cost of carbon, (ii) market price of carbon, and (ii) the social value of carbon.

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Social Cost of Carbon (SCC) is an estimate of the economic damages associated with a small increase in carbon dioxide ($CO_2$) emissions in a given year

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The Social Cost of Carbon (SCC) is the most common approach to carbon valuation. SCC is a monetary indicator measuring the present value of the global damage caused by an additional tonne of green-house gasses (GHG) emitted into the atmosphere. The SCC is often used in cost-benefit analysis to measure the value of the avoided damages, and thus the benefit of a mitigation project.

The SCC can be applied to estimate the economically optimal level of pollution, which, as in many other cases of environmental pollution, is most likely not equal to zero. Figure 13 shows an economically optimal level of abatement measures, based on SCC and marginal abatement costs, would be defined. The marginal abatement costs (MAC) are rising as the pollution level is decreasing. At the same time, the higher the pollution level, the higher the SCC. Thus, the optimal level of pollution is found where the MCA equals the SCC, which is at pollution level q and abatement costs p. This means, that the costs for removing an additional tonne of GHG from the atmosphere are equal to the global economic damage this tonne of GHG causes.

The SCC is generally estimated by employing an integrated assessment model. This model combines a diverse body of information relating to economic growth assumptions, carbon emission forecasts, abatement cost estimates and global warming damage functions. In these models, impacts at different times in the future are estimated and discounted back to present values to find the damage of a tonne of greenhouse gases emitted into the atmosphere.

There are no internationally agreed standards or approaches on how to estimate the SCC. Therefore, the values differ significantly between sectors, countries and over time. When comparing 47 studies with 232 estimates of social costs of carbon, the estimated

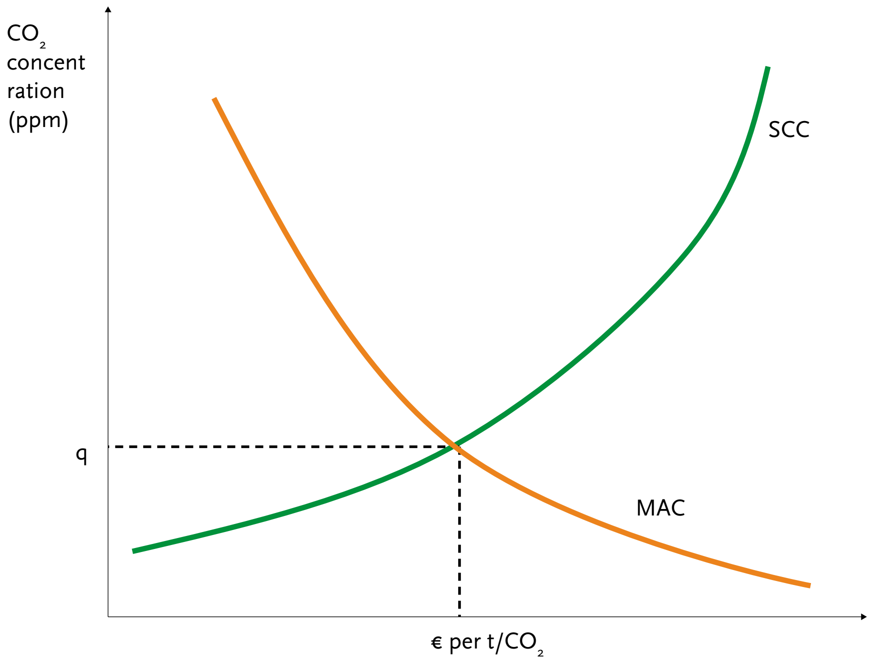


**Figure 13.** The optimal level of pollution.

mean SCC was 49 € per tonne of $CO_2$. However, the estimates were ranging from -4.8 to 1,777.5 € per tonne of $CO_2$.

These considerable differences result from the diversity of applied methodologies, models and underlying assumptions. They reflect also the uncertainties in the estimation of the integrated models. These uncertainties are related to scientific, economic and ethical assumptions used, like population and economic growth projections, the damages associated to climate change, the selection of the discount rate, methods used for the valuation of non-market goods and services and many other things. A key variable in calculating the social cost of carbon is the "discount rate." The discount rate reflects the challenge of capturing the time factor in climate policy. It contains three assumptions, which are: (i) that humans prefer to receive benefits in the present rather than the future; (ii) that future generations will be richer, thus a monetary unit (e.g. euro) will be worth less to them; and (iii) the opportunity cost of capital (that there are a variety of investment options for any given sum of money). The choice of discount rate influences whether cost benefit analyses would recommend investing in greenhouse gas reductions today or much later. From this perspective, the higher the discount rate, the less significant future costs and benefits become.

## The market price of carbon reflects the value of traded carbon emissions.

Market prices of tradable carbon emission rights or carbon credits can also be considered to measure the value of carbon sequestration. Carbon credit prices reflect quality differences, supply and demand conditions, institutional factors and transaction costs. There are two main types of carbon markets, compliance and voluntary markets.

In compliance markets carbon credits associated with national or international regulatory frameworks are traded.

Compliance markets cover transactions associated with national or international regulatory frameworks limiting the greenhouse gas emissions. In these markets only specific compliance credits can be traded. The two biggest compliance markets are the European Union Emissions Trading System (EU ETS) and the Clean Development Mechanism (CDM) in the frame of the Kyoto Protocol.

The EU ETS works on the 'cap and trade' principle. A 'cap', or limit, is set on the total amount of certain greenhouse gases that can be emitted by the factories, power plants and other installations covered by the regulatory system. The cap is reduced over time so that total emissions fall. In 2020, emissions from sectors covered by the EU ETS will be 21% lower than in 2005. Within the cap, companies receive or buy emission allowances, which they can trade with one another as needed. They can also buy limited amounts of international credits from emission-saving projects around the world.

Under the Kyoto protocol some countries (Annex B Parties) have accepted targets for limiting or reducing emissions. These targets are expressed as levels of allowed emissions over the commitment period. Countries can reach their targets by reducing their own emissions or by setting off emissions by other means (compliance credits). Thus, the framework foresees four main types of compliance credits: (i) Assigned Amount Units (AAU) which can be traded among countries that agreed legally-binding emissions reductions. Emissions trading, as set out in Article 17 of the Kyoto Protocol, allows countries that have emission units permitted them but not "used" to sell this excess capacity to countries that are over their targets; (ii) Certified Emissions Reductions (CERs) allow countries with an emission-reduction or emission-limitation commitment under the Kyoto Protocol (listed in Annex B of the Kyoto Protocol) to offset part of their emissions reduction targets through investment in developing countries; (iii) Emission reduction units (ERU) also known as "joint implementation," defined in Article 6 of the Kyoto Protocol, allows a country with an emission reduction or limitation commitment under the Kyoto Protocol (Annex B Party) to earn emission reduction from an emission-reduction or emission removal project in another Annex B Party, which can be counted towards meeting its Kyoto target; and (iv) Removal units (RMU) on the basis of land use, land-use change and forestry (LULUCF) activities. These activities include forest-related activities such as reducing emissions from deforestation and degradation, enhancing the sequestration rate in new or existing forests, and using wood fuels and wood products as substitutes for fossil fuels and more energy-intensive materials. A variety of options for the mitigation of GHG emissions also exists in other land use systems. The most prominent example is agriculture, where options include improved crop and grazing land management (e.g. improved agronomic practices, nutrient use, and tillage and residue management), restoration of organic soils that are drained for crop production, and restoration of degraded lands.

In voluntary markets carbon credits, which are outside the compliance markets, are provided to private businesses and individuals.

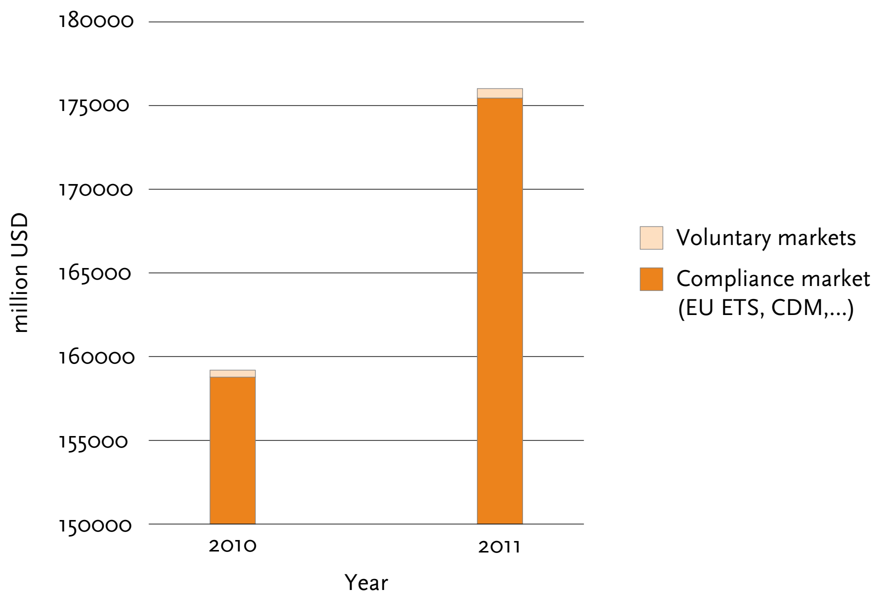


**Figure 14.** Total trade of carbon in 2010 and 2011 (data source: WB 2013).

Voluntary carbon markets provide carbon credits to business and individuals seeking to reduce their carbon footprint. In this markets encompass all carbon units, which cannot be traded in compliance markets. One important issue for these markets is that their standards for evaluating and tee important issue for these markets is that their standards for evaluating and monitoring greenhouse gas reduction projects are typically less stringent than on compliance markets. The advantage of less strict standards is lower development/transaction cost, which makes the voluntary market especially attractive for small and sustainable projects. On the other side, weaker standards could also lead to certification of projects that do not provide their stated benefits.

There are three main categories of voluntary carbon units: (i) Verified Emissions Reductions (VER) (also referred to as over-the-counter voluntary offsets or OTC) , which include project-based carbon units verified through a voluntary certification process by a third party, the project developer , or carbon unit provider; (ii) Carbon Financial Instruments (CFI) which are allowance based carbon units created under the Chicago Climate Exchange (CCX) and European Climate Exchange (ECX) voluntary cap-and-trade scheme.

Globally, carbon markets are the largest class of environmental or emissions trading markets, in terms of volume and market value. The major part of the global trade is conducted on compliance markets. For example, in 2011 the total amount of transactions in carbon markets was worth around 176,020 million USD of which 99.7% were done in the compliance market and 0.3% in voluntary markets (Figure 14). The vast majority of the compliance market transactions were done on the EU ETS and the CDM markets.

The Social Value of Carbon Sequestration reflects the peoples' willingness to pay to reduce the quantity of carbon in the atmosphere.

Another approach to assess the social value of the carbon sequestrated is by estimating the willingness to pay (WTP) of individuals to reduce the quantity of carbon in the atmosphere. This is done with methods that relay on questionnaires in which we ask respondents to indicate their WTP for the valued ecosystem service.

There are a number of studies that has attempted to determine the social value of carbon sequestration in forests. For example, in 2007 a Spanish study explored the population's willingness to pay for the implementation of an afforestation programme. This programme would also contribute to $CO_2$ reduction in the atmosphere. The estimated value was 0.00002 € per t of $CO_2$ per person and year. A similar approach was taken some by another Spanish study, where also the contribution of afforested marginal agricultural land to social welfare was assessed. However, the estimated WTP for $CO_2$ was between 20 and 1,250 times higher and ranged from 0.0004 to 0.025€ per t of $CO_2$ per person and year. These big differences are on one hand reflecting the diversity in the preferences and knowledge of the respondents, while on the other hand they appear due to variation in valuation approaches (e.g. different valuation scenarios, detail of information provided to the respondent).

### Box 4. Social value of carbon sequestration – an example.

In the NEWFOREX project the social value of carbon sequestration (peoples' willingness to pay) was estimated by asking respondents to select their preferred option among a series of policy scenarios. These scenarios were defined by levels of provision of different ecosystem services (e.g. biodiversity enhancement, water purification, recreation possibilities and carbon sequestration) and the associated implementation costs for the respondent.

Carbon emissions are typically measured in tons of carbon or carbon dioxide. However, expressing carbon reduction in such units would clearly not be well understood by the surveyed non-expert respondents. Thus, we used the equivalent of emissions produced by a certain population group in a defined period (e.g. an average citizen in one year).

In the Spanish case study region (Catalonia) people were asked about their preferences for atmospheric $CO_2$ reduction, which would equal the annual emissions produced by a number of Catalan citizens ranging from 10,000 to 55,000 (see Figure 15). Considering that the average citizen of Catalonia emits around 7 t of $CO_2$ per year, this means that the total $CO_2$ reduction provided by forests would be in the range of 70,000 to 385,000 t of $CO_2$ per year, depending on the policy scenario. The estimations shown that the average Catalonian citizen was willing to pay 0.00077 € for an additional quantity of $CO_2$, equivalent to the annual emissions of an average Catalonian citizen (7 t of $CO_2$ per year), which means 0.00011 € per each additional t of $CO_2$ reduced in the atmosphere. This result is in range with the values obtained in previous valuation studies in Spain.

Despite calculating a mean WTP value for carbon sequestration, WTP varies among the population depending on respondents' socio-demographic characteristics, like education level, gender and place of residence. Nevertheless, the surveyed population considered that it is vital to reduce carbon in the atmosphere, and that forests play an important role in mitigating atmospheric $CO_2$, although, in combination with other measures, like reduction of industrial and transportation emissions.

| Quantity of carbon stored – equivalent to the emissions of … | | | |
|---|---|---|---|
| current situation | alternative 1 | alternative 2 | alternative 3 |
| **10,000** Catalonian citizens | **25,000** Catalonian citizens | **40,000** Catalonian citizens | **55,000** Catalonian citizens |

**Figure 15.** Graphical presentation of carbon sequestration alternatives, used in the Catalonian case study.

Carbon valuation approaches are based on different assumptions and differ in what is being valued.

The three ways of valuing carbon differs in exactly what is being valued and according to whether a societal or market perspective is taken. SCC considers the social perspective and is based on future damage, and is the one requiring the largest amount of calculus and with the largest set of assumptions. Carbon valued through markets relies on different types of abatement measures. Typically it is assumed that markets are the most efficient instrument to allocate mitigation measures across agents, where the price is established based on demand and supply of carbon emission rights. However, this might not be the case when carbon markets are strongly regulated and the "products" sold are not standardized. Finally, valuation through estimation of WTP relies on respondents preferences for carbon reduction in the atmosphere. These preferences are assumed to be based upon their understanding of the risks (e.g. environmental, social and economic impacts) related to the increased carbon levels in the atmosphere. Clearly, it remains an issue to what degree the average citizen is correctly informed about future changes or if the level of insight and information is questionable, thereby resulting in potential problems of using this method to estimate the value of carbon sequestration. Nevertheless, it gives a good indication of the distributional aspects – in terms of where to do activities to reduce carbon, and who benefits from it.

Estimates of the social costs and value of carbon are sensitive to the underlying methodology and assumptions adopted. They are also affected by discount rates applied which help determine whether present values of future carbon benefits rise, remain constant, or fall over time. In the absence of a globally agreed methodology, the social based approaches are considered appropriate in appraising public forestry carbon projects, while the market prices are more suitable in the case of private investment appraisals.

## Key messages

There are three main approaches for valuation of carbon sequestration:

1. The social cost of carbon is measuring the present value of the global damage caused by an additional tonne of green-house gasses emitted into the atmosphere. Social costs of carbon are the most complex and complete valuation approach, but based on a number of assumptions and thus very dependent on the calculation methodology.
2. The market price of carbon reflects quality differences, supply and demand conditions, institutional factors and transaction cost. Carbon markets are either strongly regulated (compliance markets) or lacking established standards (voluntary markets).
3. The social value of carbon is based on individual's willingness to pay to reduce the quantity of carbon in the atmosphere. When estimating the social value of carbon, we assume that the respondent's preferences are based on a good understanding of the benefits of carbon sequestration.

The social cost of carbon and social value of carbon are more appropriate to value public carbon abatement projects, while market prices are mainly used in private investment appraisals.

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# 3.5

# Valuation of recreation, examples from case studies

**Erkki Mäntymaa, Ville Ovaskainen, Liisa Tyrväinen, Jette Bredahl Jacobsen, Bo Jellesmark Thorsen and Suzanne Elizabeth Vedel**

The recreational benefits provided by forests can be significant although they are not always reflected by market prices. Understanding their role and importance for societies is necessary for these benefits to be comprehensively recognized in decision-making. Research information concerning the benefits of forest recreation and nature based tourism is constantly ranked as a key information need in policy-making and forest planning. Measuring the values of the benefits is important in determining resource allocation decisions whether at a national or regional policy level as well as at a site level. Thus, such information is also a prerequisite for sustainable decision-making concerning the management of the forest environment.

Research related to economic valuation methods and values of environmental and recreational benefits started in Europe in the late 1970s. Many studies of the recreation values of recreation sites and protected areas have applied the travel cost method that uses the cost of travel as a proxy for the price variable to estimate the recreation demand curve. Zandersen and Tol's (2009) review of 26 European travel cost studies of forest recreation from 1979–2001 indicates a substantial variance in the estimates of forest recreation values across studies, ranging from 0.66 € to 112 € per trip (in 2000 euros). Different geographical locations, the supply of the benefits, i.e. the distance to recreational areas and cultural differences, along with the long time period of the reviewed studies, have affected the differences of the benefit estimates.

Recently, stated preference methods such as contingent valuation and choice experiments have been increasingly used to overcome the limitations of the travel cost method as regards the measurement of non-use values and potential changes or future projects related to recreational possibilities. Lindhjem (2007) made a meta-analysis of 28 stated-preference studies of non-timber forest benefits from Northern Europe (Finland, Sweden and Norway) in 1985–2005. The types of benefits valued were forest protection, multiple use forestry, forest biodiversity or other benefits including tourism to forests meaning that only a part of the studies analysed the values of the recreational benefits of forests. Also according to this review, willingness-to-pay (WTP) estimates varied substantially with the type of good, geographical scope and details of the valuation method. As one of the most important conclusions, the insensitivity of the amount of WTP to the size of the forest questioned the use of simplified WTP per area measures for complex environmental goods.

As mentioned earlier in this volume, nature-based recreation can be divided into two basic categories. First, recreational trips of a few hours to a single day are made as part of everyday life near the sites where people live, usually in or nearby urban areas. The second type of recreation is longer holiday trips to more remote special sites constructed and/or reserved for recreation and mostly requiring an overnight stay at or near the site. The latter type is also often called nature based tourism. Naturally, the value of a longer holiday trip can be substantially higher than that of a single daytrip. The case studies of the NEWFOREX project cover both types of forest recreation, i.e. short visits and holiday trips.

## Valuation of recreational benefits from daytrips – the Danish case

The Danish case study is an example of valuing the benefits mainly related to the first type of recreation, i.e. daytrips, in Denmark, one of the most urbanised regions in Europe. Here, most of the forest areas are privately owned and these areas are an important foundation for creating habitats, which support a diverse flora and fauna as well as areas for recreation options for the a growing urban population. The Danish landscape is characterized by intensive agricultural production, and forest areas therefore have a central role in providing these recreational opportunities and space for leisure time activities for the public. Studies have shown that the Danish forests receive 75 million visitors each year, and that these visitors are mostly seeking peace and quiet surroundings.

Politically there are many ways to try to influence the management of these forest areas to enhance the recreational opportunities; one way is to allow the public access outside roads and paths in private forests. Currently, access in private forests is only allowed on forest roads and paths. A widening of this access right has been discussed regularly in Danish media and forest policy processes in the 'Anemone-rule', which if implemented would imply public access to the forest floor up to 15 meters from roads and paths. Inspired by this debate, the Danish case study has investigated the public's valuation of enhanced public access in privately owned forests.

The data for the Danish case study were collected through a national online survey in 2011. The questionnaire included information on the case study area and the environmental values which could be affected by forest management changes, along with questions regarding household consumption patterns and socio-economic information of the respondent. A choice experiment was used to elicit preferences for *increased access* to forests with three alternative levels: the current access rights, access outside roads and paths allowed on 50% of the area, and access on foot allowed everywhere on the forest floor. Currently, the public has access everywhere on the forest floor in publicly owned forests, and they make up 25% of the total forest area in Denmark, cf. Table 10.

In the Danish case study, the mean WTP for access outside roads and paths was quite similar whether allowed on 50% or 100 % of the area, so both cases can be considered as one change from the current practice with access on 25% of the area up to 50–100% 2014). It turned out that people had very diverse preferences for how much they valued this increased access rights in the forest – some were willing to pay for increased access whereas others believed it would be harmful for the nature and wildlife.

The Danish population was remarkably split on this issue: One group (accounting for some 51% of respondents) had a negative mean WTP of –42 €/year, whereas the mean WTP of the other half (49%) of the population was positive and as high as 70 €/year (see Figure 16). The average WTP across all respondents in these two groups is positive, but this average figure conceals very different opinions among the respondents. The

Table 10. Recreation-related attributes considered in the Danish and Finnish case studies.

| Recreation-related attributes | Current practice | Proposed alternative practices | |
|---|---|---|---|
| **Danish case** | | | |
| Access on foot outside roads and paths | Access on road and path and on 25% of the area also outside road and path | Access outside road and path allowed on 50% of the area | Access outside road and path allowed on 100% of the area |
| **Finnish case** | | | |
| Outdoor routes in private forests | No change: 100 km of routes | Decrease: 80 km of routes | Increase: 120 km of routes |
| Quality of the landscape: traces of intensive forestry operations | No change: visible on 20% of the sides of routes | Slightly improved: visible on 10% of the sides of routes | Clearly improved: not visible along the sides of routes |

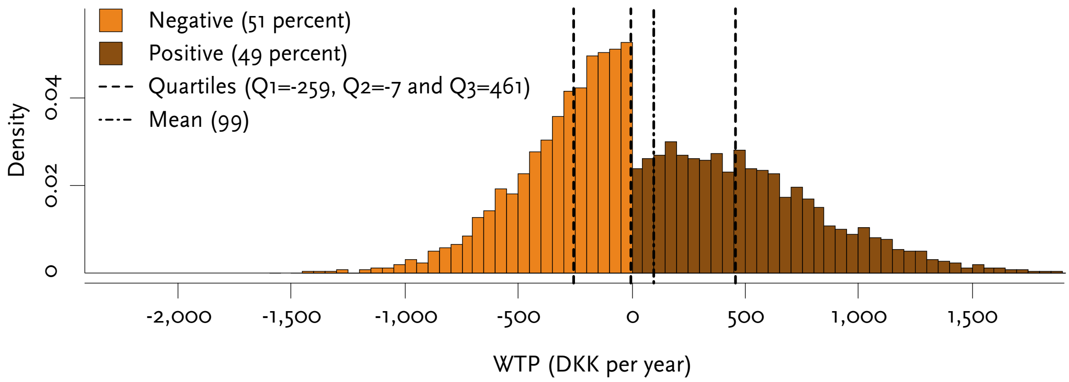


Figure 16. Illustration of how the Danish population is split in their view upon increased access rights on private forest land. A small majority (the red 51%) has a negative WTP for such a change, seeing it as equivalent to a loss in recreational quality equaling mostly somewhere between 0 and 1,000 DKK/year. A large minority (green 49%) has a positive view and a WTP mostly between 0 and 1,500/year.

reasons for the pattern in this case appear to be differences in attitudes running across most groups of the population regarding the issue of protection versus use of the environment. People concerned with the protection of biodiversity, the quality of recreational experiences etc. tended to belong to the group with negative WTP, whereas people focused on uses and less concerned about protection tended to have a positive WTP. Clearly, insights like this will change and inform the policy debate.

Thus, an important aspect of environmental valuation is that, like in the market place, people 'vote' with a weight corresponding to their WTP for the good. If we only look at the average welfare effect of a specific environmental change, we overlook the distributional effects between people, i.e., how many and who are the winners and how many and who are the losers from the proposed change. It may be that a small group of people with very high WTP for an environmental change outweighs the majority of the population who would be worse off if the change is implemented.

The example in Figure 16 shows how advanced valuation methods can uncover preference variation in the public and go beyond assessing only the aggregate values.

## Valuation of recreational benefits from nature tourism – a Finnish case

As nature-based tourism is mostly based on tourists' outdoor activities, the natural environment should be attractive and meet the tourists' expectations. In Finland, it is usual that different management practices of forests have an impact on how pleasant and enjoyable tourists see the quality of the environment. This matter is particularly pronounced in areas where privately owned lands dominate. In privately owned forests, the landowners do not necessarily take into account the quality of the environment and its impact on tourism when making decisions on forest management. In order to improve the co-ordination of the tourism industry and commercial forestry the Finnish case study analysed visitors' valuations of the recreational and environmental benefits of forests.

The Finnish case study was conducted as a choice experiment during the winter-spring and summer-fall seasons in 2011. The data were collected from 1100 tourists and local visitors as an on-site guided survey in the Ruka-Kuusamo area, a major winter sports and nature tourism centre in Northern Finland. The questionnaire was written in Finnish, English, French, German and Russian. About 21% of the respondents were other than Finnish speaking in line with the estimated annual share of foreign visitors. The survey asked about the preferences for two recreation-related attributes: the *total length of outdoor routes* in the private forests of the area, and the *quality of the landscape* as illustrated by the frequency of visible traces of intensive forestry operations, especially clear-cutting and soil preparation, along the routes. In addition, *biodiversity* was represented by the development of endangered species populations. The costs of the potential improvements of recreational possibilities were told to be covered with a *payment for environmental management* to be charged in connection with accommodation prices.

The WTP values for changes (increases or decreases) in the recreation-related and environmental attributes in monetary terms are shown in Figure 17. In the Finnish case study, the respondents were willing to pay for the improvements in the quality of the landscape. More precisely, if the quality of the landscape would be *clearly* improved so that traces of intensive forestry operations would not at all be visible along the outdoor routes, the visitors would be willing to pay more (12.17€/week) than if the quality would only increase *slightly* (10.82 €/week). On the other hand, the WTP for a decrease in outdoor routes was negative (–9.99 €/week). This indicates that the visitors should be paid this sum of money in compensation for this loss of recreational benefit in order to keep them as well off as before the change. However, for an increase of the length of outdoor routes people on average are not willing to pay anything, suggesting that the supply is adequate as it is. Related to environmental benefits other than recreation, the visitor would be willing to pay for an increase in biodiversity (10.2 €/week) and claim a compensation for its decrease (36.8 €/week).

## Conclusions

The results of the Finnish case study support the idea that tourists are prepared to pay for selected improvements in the quality of outdoor recreation environments that can be accomplished through adjustments in forest management practices. The results support the expectation that one of the most important improvements in the landscape for tourism would be mitigating the negative effects of final fellings and regeneration. In

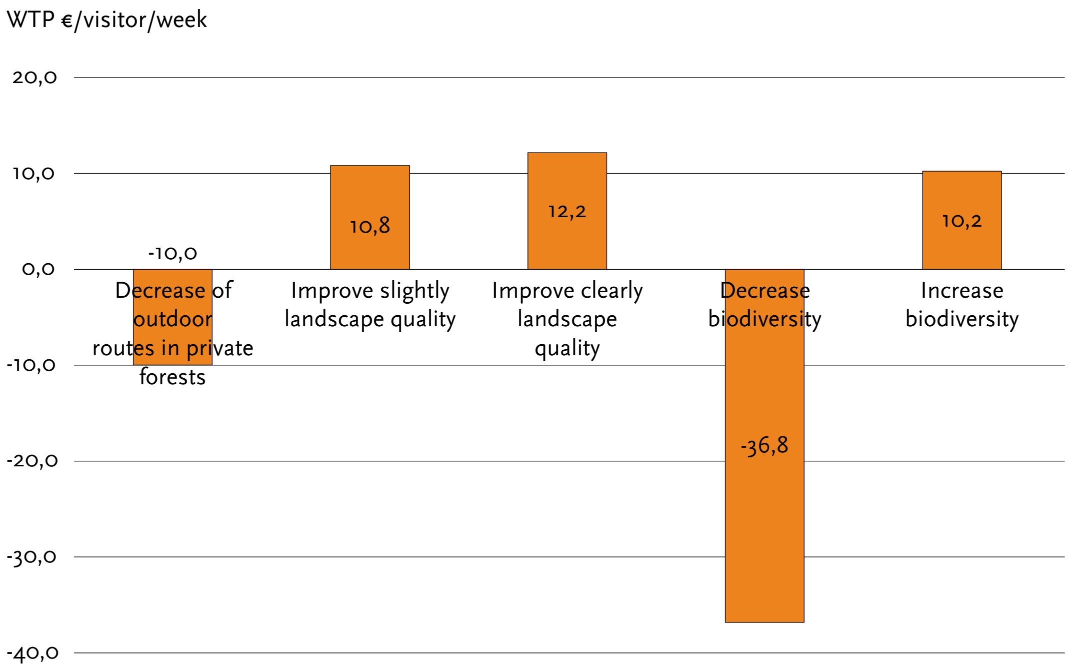


**Figure 17.** Willingness to pay for changes in recreational and environmental quality among the visitors of the Ruka-Kuusamo nature tourism area in Finland.

terms of required changes in forest management practices, this suggests using delayed fellings and avoiding clear-cutting and intensive site preparation along trails and resting places, for example. In such scenically sensitive areas selective harvesting and natural regeneration through small patches would be used instead.

The Danish case draws attention to the apparently plausible expectation that enhanced access to an area would give benefits for all people. The results strongly suggest that such an assumption overlooks two effects that are possibly quite important in the Danish case where forest land is used heavily for recreation and, at the same time, constitutes an important habitat for biodiversity conservation. First, some recreationalist groups may experience widespread rivalry and congestion decreasing the quality of their recreational experience and thus they perhaps see increased access mainly as increased pressure on a common pool resource. Second, other groups may worry about the effects on biodiversity and habitats, and hence factor in these as externalities of increased access rights lowering their value. Such people may consider the value of any increases in access rights for all an overall negative change as they basically oppose the idea.

## Key messages

- Tourists are prepared to pay for selected improvements in the quality of outdoor recreation environments that can be accomplished through adjustments in forest management practices.
- In scenically sensitive areas selective harvesting and natural regeneration through small patches should be used instead of clear-cutting and intensive site preparation.
- People's preferences related to recreational values vary a lot. In areas with intense recreational use many find that increased access may be harmful for nature and wildlife in the forest.
- In the areas of intensive recreational use some people may experience widespread rivalry and congestion decreasing the quality of their recreational experience and may see increased access mainly as increased pressure on a common pool resource.
- Advanced environmental valuation techniques can provide policy makers with politically crucial insights into winners and looser from environmental policies.

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3.6

# Valuation of biodiversity, examples from case studies

**Jette Bredahl Jacobsen and Anna Bartczak**

Biodiversity and its loss has been one of the major topics in environmental policy for the last two decades, currently highlighted by the Intergovernmental Platform on Biodiversity and Ecosystem Services (www.ipbes.net), which is an attempt to ensure the same long-run and enduring policy commitment to combatting habitat and biodiversity losses as the Koyoto process and in particular IPCC has been for the climate change policy agenda. The concerns are also reflected in an increasing number of articles estimating monetary values of case specific biodiversity changes and protection. Biodiversity is often defined as the variety of all forms of life. It can refer to genetic, species, habitat, or landscape variation. Because of this scale specificity all value estimates are also context specific. How biodiversity is valued, depends on which level of variation we are looking at, how it is quantified, and which services of biodiversity are in focus. Studies estimating the value of biodiversity protection measures or the cost of biodiversity losses are usually based on individuals' stated preferences.

To explain the biodiversity concept to respondents in a questionnaire is a challenging task. It is crucial to present biodiversity changes in a simple way – capturing the core values believed relevant by the people in focus and at the same time being precise. Running several focus groups is a key instrument to help frame the biodiversity definition in a comprehensive way and allow testing respondents' understanding of this term.

Many studies addressing biodiversity protection has targeted the valuation of habitat protection – for example enhancing the size of nature protection areas. Others look at the conservation of endangered species. While the conservation of an endangered species in a specific area may not enhance biodiversity much at the specific site (only by one species), it is the ultimate measure to increase biodiversity at a landscape level. This, together with the fact that it is easy to communicate, may explain the large focus on endangered species in the biodiversity valuation literature. More recently some studies looked at the functionality of biodiversity. Qualitative studies show that people do not necessarily distinguish between the functionality and the value of biodiversity per se (i.e. the existence value). Therefore, measures that can capture the functional values, like structural forms, may be used for valuing biodiversity by the general public.

Finally, some studies look more into the species diversity at a specific location, whereby it is not as such conservation of endangered species that will increase biodiversity, but also presence of species common at a larger scale. Typical measures used for that will be a number of present species.

In recent years, it has become popular, to estimate the willingness to pay (WTP) for changes in biodiversity protection levels by the use of a choice experiment. Choice

Figure 18. Obtaining the value of biodiversity improvements may take a long time. A natural cycle may take several centuries. Drawing: Henrik Meilby.

experiment is a stated preference method, where respondents are asked in a questionnaire to make trade-offs between policies varying in the implied cost as well as the attributes describing the ecosystem services. Consequently biodiversity may be valued as a component in a policy changing biodiversity levels, recreational opportunities and in some cases also other attributes.

In the literature, biodiversity has been valued in different case study contexts, and almost inevitable result of the ecosystem focus embedded in conservation. The case studies we present here share that feature, but they differ in terms of forest location, characteristics, ownership rights and scale. Because of that it would be very hard to apply a single biodiversity definition and a measurement in all case studies. The number of species protected has been chosen as a way to describe biodiversity in several of the cases. The reason is that it is simple to communicate, easy to quantify, it has management relevance, and it may be seen as a representation of biodiversity or species in general. In two cases studies the attention was focused on the naturalness of forest ecosystems processes (in the other words decreasing the role of human intervention in those processes). Additionally, to test people's loss aversion i.e. to check if losing a species is worse for them than improving the conditions, potentially preventing future losses, in two cases both an increase and a decrease of biodiversity as opposed to the level today have been valued.

## Illustrative case study results

In the Finnish case visitors were asked to value forest policies in a tourism area, trading off different conservation intensities against recreational initiatives and a money contribution. They were willing to pay 40 € per visitor and week to avoid 10% of the species present today going extinct locally in the area. In the other direction, visitors were willing to pay 10 € per visitor and week for a 10% population increase for existing species. In the Italian case, we found a WTP to avoid reduction in the number of abundant species (-25 species) of 36 €/household and year, whereas there was no willingness to pay to increase the number of species present.

Table 11. Biodiversity attributes used in the NEWFOREX project – their levels and WTP.

| Attribute definition (specification?) | | Attribute levels (without SQ) | WTP for biodiversity changes per household/year | Case study site | Case study region |
|---|---|---|---|---|---|
| Number of species | Fauna | -_25<br>+ 10 | 0.62 €<br>per species | Forests in a region | Italy |
| | Fauna and flora (endangered) | - 10% extinct<br>+10% population increase | 40 €<br>10 €* | Forests in a region | Finland |
| | | 50<br>100 (secured survival) | 1.9 €<br>per species | Forests in a region | Denmark |
| | Flora | 3<br>5<br>7 | 12 €**<br>per tree species | Forests in a region | Catalonia |
| Naturalness of forest process | | Low level<br>Medium level<br>High level | 104 €<br>123 €<br>160 € | Forests in a region | Denmark |
| | | High level<br>for a commercial part of the forest (in 250 years)<br>for a second growth of the forest (in 150 years) | 8 €<br>10 € | Single forest | Poland |

Note: * per visitor/week; ** per person/year

In the Catalan case, biodiversity was explained as a consequence of how many tree species was present. Thus it was explained to respondents that policies increasing tree species diversity will increase the overall biodiversity. We found that respondents were willing to pay 12€ per person and year per extra tree species present – up to 7 species.

In the Danish case we valued the presence of species and natural dynamics as separate attributes. For the species protection, we found a WTP of 1.9 € per household and year for each extra endangered species conserved, with the levels of 50 and 100 species being presented. Earlier studies have shown a WTP in the same range per species. This shows that the marginal utility of an extra species is not reduced significantly within a range of up to 100 species in the Danish case.

The in the Danish case, the "naturalness" or protection of ecosystem functions important for biodiversity was represented by the presence or not of specific natural processes. It was explained to respondents that trees (up to 5 per hectare) could be left for natural aging and decay in otherwise intensive managed forest, which would increase the potential for natural decomposition processes to occur and benefit diversity. The WTP for this was 104 € per household and year. If instead the area of untouched forest was increased to 7% in the case region, the options for natural processes would also be enhanced. Respondents were willing to pay 123 € per household and year for setting aside 7% of the forest area. Combining the two, resulted in a WTP of 160 € per household and year. It is expected that the WTP for both initiatives together is smaller than the sum of the two, as there is likely to be a diminishing marginal utility.

The Polish case also represented biodiversity by the “naturalness” and as for the Danish case it was closely linked with management practices. Thus the valued initiatives were second growth forests and commercial forests, resulting in a high biodiversity level in 150 years and 250 years respectively. The new approach for valuing biodiversity here lies in an acknowledgement of the fact that increasing biodiversity is a slow process that may go way beyond present human beings life. Explaining to respondents that policies being initiated now will first have consequences far into the future is challenging. Nevertheless it was proven possible and we found a WTP of respectively 8 and 10 € per household and year. Table 11 gives an overview of the biodiversity attributes and the estimated WTP.

## Conclusions

In general the WTP’s are quite high, especially when comparing with the other ecosystem services being valued. This is in line with what many other studies find, namely that biodiversity protection is the highest valued ecosystem service from forests. Therefore it is important to include these values, even if they are less tangible than many of the others and to a large extend consists of non-use values. We also see, as expected, that a loss in biodiversity is (dis)valued more than an increase.

The other thing to notice is the use of descriptive measures of biodiversity like ‘naturalness’ or natural processes. These seems quite important to people, but are not often used in valuation studies. While important, they become very context specific, and therefore difficult to compare between countries. Numbers, like species preserved, are more comparable. But again, caution has to be given to the exact context.

### Key messages

- Biodiversity underpins a lot of other ecosystem functions, but valuation studies often focus on habitat or species conservation.
- Valuing endangered species and biodiversity contains many moral and ethical aspects, possibly causing an upward bias on value estimates.
- General perceptions of nature and environment varies a lot across people, and people that are more environmentally concerned place higher economic values on the protection of biodiversity.

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