What Science
Can Tell Us

The Provision of Forest Ecosystem Services

Volume II: Assessing cost of provision and designing economic instruments for 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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Authors

Jens Abildtrup, Institut National de la Recherche Agronomique, France
Anssi Ahtikoski, Finnish Forest Research Institute, Finland
Elena Górriz, Centre Tecnològic Forestal de Catalunya, Spain
Paola Gatto, University of Padova, Italy
Udo Mantau, University of Hamburg
Robert Mavsar, European Forest Institute, Finland
Erkki Mäntymaa, Finnish Forest Research Institute, Finland
Ville Ovaskainen, Finnish Forest Research Institute, Finland
Davide Pettenella, University of Padova, Italy
Claudio Petucco, Institut National de la Recherche Agronomique, France
Irina Prokofieva, Centre Tecnològic Forestal de Catalunya, Spain
Florian Schubert, Universuty of Hamburg
Anne Stenger, Institut National de la Recherche Agronomique, France
Bo Jellesmark Thorsen, University of Copenhagen, Denmark
Suzanne Elizabeth Vedel, University of Copenhagen, Denmark
Sven Wunder, Centre for International Forest Research, Brazil

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 nonmarket 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.

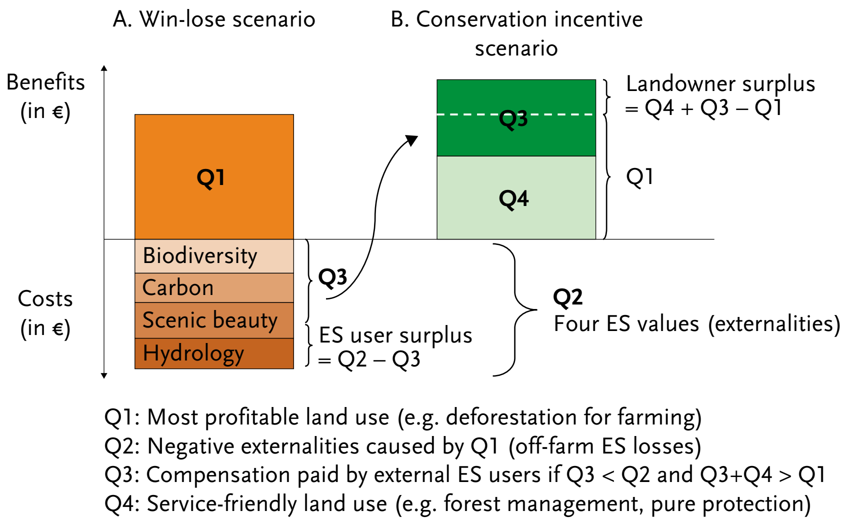

Figure 1. Externalities in ecosystem services provision. Modified from Pagiola and Platais (2007).

(*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/).

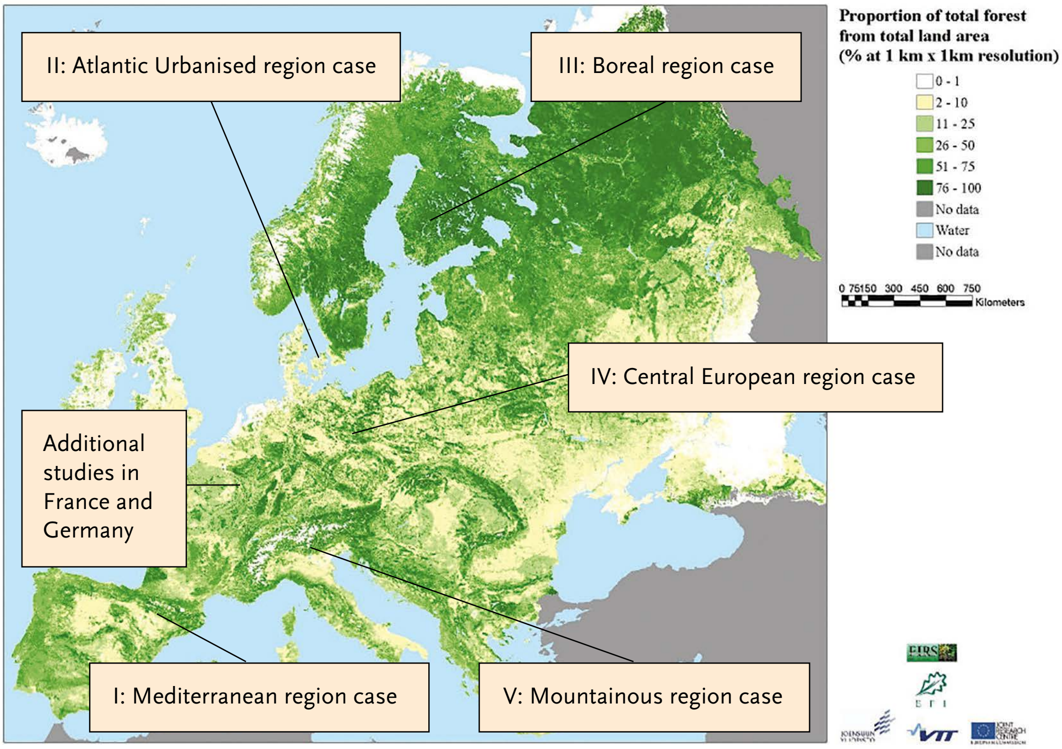

Figure 2. The five European case studies in NEWFOREX, and the supporting French and German forest owner studies. The developing country case in Brazil is not shown.

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 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 this 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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Pagiola, S. and Platais, G. 2007. Payments for environmental services: from theory to practice. Washington DC: World Bank.

TEEB 2010a. The economics of ecosystem and biodiversity: mainstreaming the economics of nature: A synthesis of the approach, conclusions and recommendations of TEEB. www.TEEBweb.org

TEEB 2010b. The economics of ecosystem and biodiversity. Report for business – executive summary. www.TEEBweb.org

2.

Linking cost of provision and policy instruments

2.1 Why do we need to know costs of provision to design economic instruments?

Anne Stenger, Irina Prokofieva, Paola Gatto and Davide Pettenella

Identifying the marginal costs of providing non-marketed ecosystem services is one essential part of the standard economic framework of social decision-making on ecosystem services provision. The knowledge of marginal social benefits (see Volume 1 for further information) informs the social goals for ecosystem service provision and the choice of the appropriate policy instruments. These aspects are the three other essential parts.

Assessment of the costs of provision of forest ecosystem services is important for forest management and especially for designing public policy instruments for enhanced provision of ecosystem services. This knowledge is necessary to achieve cost-efficiency in the use of public funds when designing voluntary or mandatory economic instruments for ecosystem, or to reduce private and public costs as far as possible when implementing e.g. mandatory management standards.

In this volume we focus on the assessment of costs of provision and on the design and implementation of in particular economic policy instruments. Managing forests for multiple purposes like non-marketed ecosystem services creates real stakes, especially if incentives are proposed to forest owners who are expected to change their management to provide more environmental services.

As Europe is characterized by large regional differences in the natural conditions (e.g. various climates), institutional contexts, and public demands for forest ecosystem services, there is a need to apply a variety of the cost assessment methods to reflect this diversity. Accurately estimating costs associated with environmental service provision is a challenging task, especially in forestry where some costs can be uncaptured or inaccurately captured for different reasons:

- spatial heterogeneity: extrapolating cost estimates from several stands to a larger forest scale;
- time horizon: the potential lags between the change in forest management and the observed environmental service provision

- uncertainty: while management changes can be implemented, it is often not guaranteed that they will in fact result in the environmental improvement aimed for, e.g. due to other natural risks.

Assessing the costs of provision in any specific context is a complex endeavour that requires an answer to two questions:

1. What is being measured in terms of management change and environmental change?
2. How can we assess the cost of the management change?

The cost structure or the total costs comprise both direct costs of undertaking a specific forest management action (like choosing more expensive harvest methods) and indirect costs (opportunity costs like income foregone, transaction costs and feedback costs), in other words all the categories of expenses that a forest owner bears to provide an environmental service. Assessing these costs requires that we understand the different factors – technical, social and economic – that affect the costs of the specific management changes and the resulting change in environmental service provision. In addition to the identification of cost types and components, the analyst need to identify the specific measures (e.g. changes in forest management activities) needed by forest owners to ensure the specific increase in provision of environmental services we pursue. This implies that the links between the supply of ecosystem services and the costs are quantified.

The second question addresses the choice of an appropriate approach to assess cost of provision. This choice is often constrained by data availability and accessibility (i.e. markets prices, growth functions, management alternatives) or the difficulty and costs associated with direct observation of samples of forest owners who actively manage their forest in different ways.

To measure ecosystem service provision, the choice of a relevant indicator is a crucial task, as it requires the indicator to be reliable and credible for both the supplier, that is the forest owners, and the beneficiaries, the users paying or more often perhaps the general public funding the measures. Moreover, defining measures for ecosystem services quantification is a challenging task in itself. Sometimes the challenge arises from the large number of different indicators available – in other cases – from the lack of relevant indicators. In the absence of direct measures of ecosystem services, indicator proxies can be used, but they give only a general and imprecise idea about the change in provision.

Therefore, instead of basing cost of provision estimates and policy instrument design on good quantitative measures of provision of ecosystem services, they are often based instead on easier to observe variations in management practices, e.g. by identifying some relevant actions to be undertaken that departures from what is considered best practice or Business as Usual (BAU).

A final complication for the cost assessment and policy design is that a particular action may have an impact on more than one ecosystem service, raising the issue of feedback costs and requiring the analyst to take into account the aspects of joint production. In other cases, in turn, several different management actions may be required to obtain an increase in a specific ecosystem service.

Two main approaches for cost assessment have been used in research as well as practice: The first one comprises technological or engineering approaches focusing more on outcomes in terms of management changes and measurable production, the second one on behavioural approaches focusing on forest owners management decisions. In the engineering approach, the opportunity costs of forest ecosystem services are computed as the difference in the present value of net income from forestry between the management regime enhancing the provision (management alternative) and the conventional regime (BAU). In the behavioral approach, cost assessment is done related to hypothetical management alternatives. To evaluate new management actions, public decision-makers can rely on forest owners' own assessment of their expected costs. This "stated-cost" approach is often based on surveys where forest owners are asked directly to state their costs, or to choose between different scenarios defined by specific voluntary management activities and restrictions linked to monetary compensations.

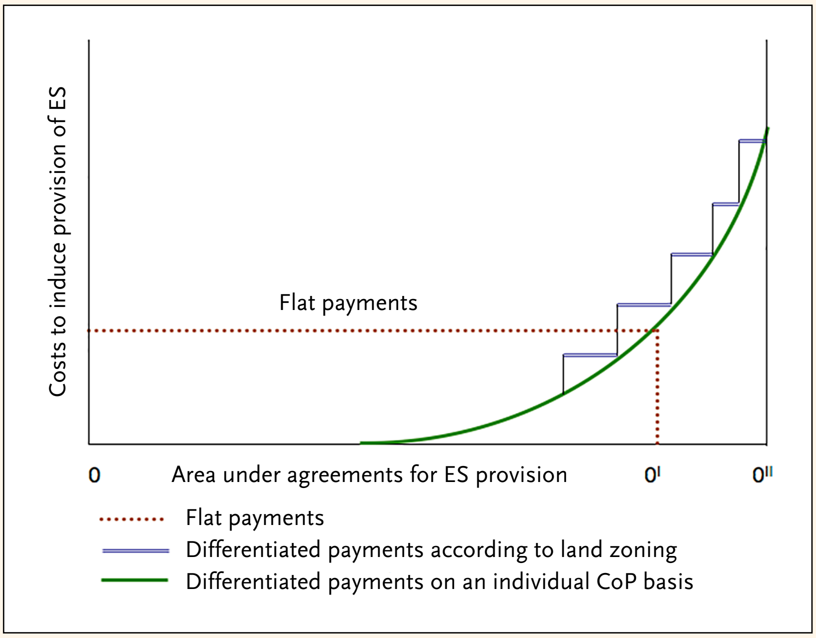
Knowledge of the cost structure of ecosystem service provision and of the underpinning technical, social and economic factors offers the basis for setting efficient targets for provision and for designing cost-effective management strategies in order to meet these targets. This is particularly relevant when developing economic instruments, where cost of provision represent the breakeven point, that is to say the minimum level of compensation forest owner would accept for undertaken the management change voluntarily.

Indeed, cost of provision measures have been, and still are, largely used in many different economic mechanisms, like for example the large public agro- or forest- environmental schemes of the Common Agricultural Policy. This diffusion of cost of provision measures in environmental policy design has happened for different reasons. First of all, to respect WTO agreements, the EU requires that environmental support measures do not include price support for being eligible under the green box of non distortive subsidies. Secondly, use of cost of provision as reference for payment may offer some advantages. Being based on 'hard' data (that is, the real costs incurred) rather than on complex estimating procedures, they might prove easier to have accepted by suppliers, tarmers anf forest owners. Furthermore, traceability and accountability of the expenditures through recorded invoices provides public support for monitoring and auditing purposes.

Cost of provision (CoP) assessments are a crucial element for guaranteeing satisfactory levels of ecosystem service supplies for given costs, and for reaching efficiency and fair impacts on equity (at the least cost for society) of public expenditures. However, some important conditions must be met. In the definition of cost of provision components a comprehensive approach must be used, that takes into account direct and indirect costs and, especially, transaction costs. The inclusion of transaction costs into policy evaluation is crucial in order to achieve the desired policy targets in an efficient way. Indeed, initiatives of economic instrument implementations often require up-front payments above ordinary direct costs of provision to cover the costs of changes and motivate actors, for example for compensating their adaptation of the management systems, or the time incurred in achieving new information and learning new practices and marketing strategies. Moreover, a fine-tuned opportunity cost approach should preferably be adopted: local or regional variation in environmental or market conditions can affect the forest owners' sets of alternative opportunities in different ways, thus creating heterogeneity in forest owner attitudes towards e.g. the different voluntary economic instruments targeting the provision of ecosystem services. A further challenge for the use and consistency of cost of provision estimates is the often significant non-homogeneity of the environmental services across space and context, which is a source of additional heterogeneity and potential inconsistency of cost of provision across agents. Therefore, a narrow approach to CoP may fail to induce a quantitative change or an upgrade in the desired service (see Box 2). This must be kept in mind in order both to achieve effectiveness through an adequate financial support and also to ensure efficiency in public spending.

Box 2. A supply curve of ES as a function of CoP and related payments.

Work published by Whitby and Saunders in 1996 has explored compensation under economic instruments. In the figure, the green curve represents a supply curve for a specific ecosystem service. Instruments based on flat levels (red horizontal dotted line in the figure) under- or over-compensate the ecosystem service providers. Differentiated payments calibrated along the step function in blue better approximate the supply curve. Payments negotiated with individual providers so as to cover the precise amount of compensation needed by each provider to accept the change (thus following exactly the supply curve as in a market transaction) can induce provision in all of the OO” area. Thus, both farmers’ rents resulting from excess payment as well as the risk of forcing provision levels with costs above benefits, leading to failure in the medium-long term, could be avoided – ideally. To achieve that, the negotiation process should approximate the payment to the marginal cost incurred by the farmers, plus the costs needed to incentivice the farmer to enter the agreement. A compromise solution between the payment to single ES providers and the flat payment could be e.g. differentiated payments according to land zoning, if that can take into account site-specificities at least to some degree.



Source: modified from Whitby, M. and Saunders, C. 1996. Estimating the supply of conservation goods in Britain: A comparison of the financial efficiency of two policy instruments. Land Economics 72: 313–25.

2.2

Quantification of management measures and ecosystem services provision

Sven Wunder, Jens Abildtrup and Bo Jellesmark Thorsen

In the design and implementation of policies enhancing ecosystem services provision two practical issues get very important. First, how is the provision measured? Measurement is important for enforcing and evaluating a given policy measure. Secondly, what measures will insure a given provision of services? Measurement and the identification of measures are in many cases two closely related questions as service provision is often measured by the implementation of specific actions. In this chapter we discuss measurement and the identification of measures (or management actions) to ensure the service provision.

Should we measure ecosystem services or actions?

Environmental policies and interventions usually target a better provision and higher value of (some pre-defined bundle of) ecosystem services. But how will decision makers be able to know to what extent they have made the hoped-for difference? In a world without transaction costs (e.g. monitoring costs) and perfect knowledge it would be preferable to base policies on a given level of provision of the service of interest. For example, a forest owner could be paid for a certain number of rare species present in her forest. This would allow the individual forest owner to choose the least costly measure to insure the demanded provision in her specific case, as the least cost measure may depend on the forest characteristics and the forest owner type. Basing the policy on the output of the management actions ensures that forest owners may choose cost-effective measures for the specific forests, provided they know the relation between action and output. This is what is often called output based regulation. However, in many situations this is not an applicable approach since measurement costs are very high and/or we do not have perfect knowledge about the relation between action and outcome. For example, it will be expensive to assess the number of species in a forests every year to obtain a measure of biodiversity provision. A second issue is to what extent we will be able to measure the actual service within our targeted time frame. There may be a long time span between implementation of a provision measure and the desired outcome. Therefore, policies of environmental service provision are often based on proxies of outcome (changes in land use) or on specific management actions (also termed input-based regulation). Furthermore, management actions are less affected by hazardous third factors (e.g. drought followed by increased non-anthropogenic fire risks). It will often be unacceptable and inoptimal to let individual forest owners carry this risk or the risk associated with lack of knowledge. However, specifying a policy based on management actions will reduce forest owners' flexibility and will typically increase their costs, as the implementing environmental principal (e.g. a government agency) would have less information than the forest owner about the cost-effective measure in individual forests.

Assessment of a baseline

Whether or not a policy is input or output based, it would always be necessary to assess a baseline: what would likely have happened to the targeted ecosystem service(s) without our intervention? And what would have been the management choice? The difference between the baseline and the actually observed service trend represents the incremental ecosystem service gained, and is often termed the "additionality" of the measure.

Normally we will look to history to construct a service baseline. For instance, if we want to avoid carbon losses from future deforestation, we have to predict a "business as usual" (no intervention) deforestation scenario, which we will often base on some simple extrapolation of past deforestation trends (e.g. the last five years' average) or a model that integrates determinants of our target land use (e.g. road building and agricultural commodity prices). Similar methods may be applied for losses of biodiversity at relevant scales. Modeling can to some extent bridge over the lack of good historical data. Hence, the reliability of our baseline also comes to depend on the quantity and quality of ecosystem service and land use data available. Note, however, that the implementing environmental principal will only be able to make qualified guesses about what actions individual agents (e.g. different landowners) would have taken under a 'business as usual' scenario without the intervention. This is usually referred to as the problem of 'asymmetric information' in measuring environmental additionality and service providers' costs. When policies have long time horizons it is important to account for potential climate and socio-economic changes which may have a significant impact on provision actions effectiveness as well as having an impact on the cost of provision.

Identification of management actions

As policies are often based on specific management actions and not directly the on the change in the ecosystem service which is the aim of the policy, it is important for the implementing principal to identify relevant and cost-effective measures. This may be a complicated task as many different measures may provide the same ecosystem service.

As an example, Table 1 reports four different management actions which have been identified as measures to increase the supply of biodiversity protection in Atlantic (deciduous) forests. In addition to a large set of potential measures affecting one ecosystem service, each measure may have impacts on more than one ecosystem service. For example, changing from conifers to broadleaved tree species in Denmark (Table 1) is normally considered to have a positive impact on biodiversity protection status in the long run. At the same time surveys have shown that most forest visitors prefer to visit forests with broadleaved species. This is an example of positive feedback between two services (joint production). If the supply of biodiversity increases the provision of recreational

Table 1. Example of how management actions (measures) may affect several services – here affecting the provision of biodiversity protection and recreational services in Denmark (the list is not exhaustive).

Examples of actions	Impact on biodiversity	Impact on recreation
Change from conifers to broadleaves	Increase in biodiversity due to more native species linked to broad-leafed forest	In general, more attractive forest for recreational users
Leaving a number of trees when doing final harvest	Increased number of insects, fungi, birds and small animals	Uncertain effect. Some users will benefit from increase in biodiversity – other will have disutility of seeing decaying trees
Leaving larger areas untouched by management	Increased number of insects, fungi, birds and small animals	Uncertain effect. Some users will benefit from increase in biodiversity – other will have disutility of seeing decaying trees and "messy" new forests
Allow public access on foot in all forest and everywhere in the forest	Potentially negative impact on biodiversity (disturbing larger animals and forest floor fauna and flora) in forests with many visitors	Increase the attractiveness of forest recreation

services increases too. However, such feedback effects may also be negative. For example, opening up for public access, i.e. increasing the recreational service this may have a negative impact on the big game population. This is an example of a negative feedback in the provision of recreational services. In Table 1 we have only considered two services. However, the considered measures may also have impacts on other services. For example, changing tree species may also have impacts on carbon sequestration and watershed protection.

Measurement of environmental service provision: Examples

In the following, we will briefly exemplify these matters for each of the four main targeted forest ecosystem services of these volumes. We refer the reader to the corresponding and more detailed chapters in Volume I of this publication.

a) Forests and watershed protection from

Hydrological ecosystem services from forests are among the most difficult to measure and predict, due to a strong variability in time and space. Important year-to-year variations in precipitation and temperatures cause natural fluctuations in service provision over time. Similarly, the complex interplay between vegetation cover and its management, soils and slopes is spatially highly heterogeneous, and long lags may occur between management changes and measurable service output (e.g. between soil restoration and stabilized water flow). Finally, many targeted watershed services refer to the mitigation of serious risks (e.g. floods, landslides, silting of water infrastructures), which by their very nature only occur in large time intervals, so that an improvement over a risk-mitigation baseline only can be reliably evaluated perhaps decades after the intervention has occurred.

Progress towards improved services is thus best measured in terms of promotion of those land uses that are likely to promote service provision (e.g. forest conservation on slopes with fragile soils). Modeling changes explicitly in space can thus under such circumstances also become quintessential, for instance using a Soil and Water Assessment Tool, to quantify the linkages between changing land management, likely service amounts provided, and the corresponding costs to service providers and benefits to service users, respectively.

b) Forests and carbon services

How do we measure changes in forest carbon stocks, and the "additionality" of any improved management? Forest carbon services – within the forest – are easier to quantify in the sense that linkages between the "action" and the "service" are more stable in time and space than for watershed services. However, there are some exceptions. For instance, forests that in normal years act as carbon sinks can during El Niño Southern Oscillation (ENSO) years become net carbon emitters. Similarly, afforestation in some high-altitude areas with fragile soils can release accumulated soil carbon to an extent that at the extreme could outweigh the gains in above-ground biomass.

As a simple accounting point of departure, forest carbon stocks depend on two factors: forest area size and carbon densities. Many assessments of forest carbon stocks thus start out from ground-based forest inventory data, supplemented by biome-wide density averages for extrapolation in space. This information they combine with remote sensing (radar, laser, or air-borne optical sensors) based estimates for area size and its changes over time.

c) Forests recreational services

Recreational services differ substantially from water and carbon services in that the end service quantity is not biophysically measurable, but instead determined by ranked human preferences for different landscape attributes. Measurements of these ecosystem services are typical based on observations in the form of frequencies, length and site of recreational visits, which form the basis for e.g. travel cost based estimates of recreational values. Forest management changes affecting recreational visits depend on the specific context of the recreational landscapes, and the preferences of the relevant populations. Forest tree ages, species, open areas, track availability, the presence of other forest guests and many other factors affect the quality of the recreational ecosystem services to the individual.

d) Forest biodiversity services

Biodiversity services are linked to a stock of forest resources (species, landscapes, etc.) that humankind derives intangible, often non-use benefits from, such as 'option' (or 'bequest') values of future generations possible yet so far unknown resource uses, and the 'existence' values current generations attribute to the survival and thrive of species that they may never see in situ. Hence, while these values may change with human preferences and technology, the underlying stock of diversity has, unlike recreational services, a well-defined biophysical dimension. However, the exact measurement of biodiversity can be cumbersome and expensive, so that proxies are also often used. This could be certain keystone species (e.g. bird diversity) or land-use proxies (e.g. near-natural forest area preserved). The latter can provide fairly good approximations of changes in biodiversity under the well-known species area relationship, predicting that habitat size is the key determinant for biodiversity levels.

Key messages

The measurement of ecosystem service provision is costly and therefore the quantification for policy measures is often based on proxies or on management actions.

- For assessment of the additionality and cost effectiveness of policies it is important to assess a business as usual scenario which can be based on historical data or/and modeling.
- Changes in the provision of a given ecosystem service can typically be achieved by several different management measures, and the most cost-effective measure should be chosen.
- The relevant measures will depend on local conditions, and the identification of relevant measures should be case or region specific.
- Management actions would typically have impacts on the provision of more than one ecosystem service. This should be considered when assessing the cost-effectiveness of measures.

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2.3

The role of private forest owners' motivations and attitudes

Elena Górriz, Suzanne Elizabeth Vedel and Anne Stenger

Diverse motivations drive private forests owners' management decisions which in turn affect the provision of various ecosystem services. By identifying owners' interests, their intentions for forest management and the constraints they perceive, policy makers are better equipped to understand and predict forest owners' behavior. Consequently, they are able to impinge on it towards the socially optimal provision of ecosystem services. Policywise, such information may be used for targeting specific policies to a subset of owners, thereby increasing cost-efficiency: e.g. when it is not necessary that all owners implement a specific change or the target population is those with lower cost of provision.

Forest owners express diverse preferences regarding the ecosystem services their properties provide. Studies of forest owner objectives and groups have categorized owners that prioritize timber (as "traditional wood producers"), those that prioritize non-timber benefits ("amenity oriented owners"), those that aim at a mixture of timber and non-timber products (as "multi-objective owners") and those that are inactive, with a rather low interest in forest management ("passive owners"). All these categories, while not being comprehensive, show that forest owners should not solely be considered as profit-maximizers, but instead as a more differentiated population of utility-maximizers with several values at stake. While this chapter refers primarily to private landowners, challenges are arising across Europe with manifold forest ownership status or decision-making structures for forest management (see Box 3).

Box 3. Typologies of forest ownership and management decisions effects.

Typically, land ownership has been equivalent to possessing the decision-making power over the forest management on that land. Across Europe we find family forests (holders being individuals), private industrial forests (belonging to private firms with industrial purposes), communal forests (owned by neighbours' assemblies), municipal forests (with e.g. a town council deciding over them) or those belonging to higher public administration levels (classically to the state, with public technicians -civil servants- or state-owned firms deciding over them) or the church. As a general rule, the first would be more oriented to short-term benefits and private goods, whereas the public ones would be more inclined to consider the provision of long-term production values and public goods. Phenomena such as urbanization and rural migration, the increase of environmental awareness, decentralization or privatization processes, or lower budgets for public forest companies are modifying the previous panorama. New agents are, hence, entering into scene: e.g. delegation of forestry decisions to forest owners' associations or consultants, NGOs acquiring land-related rights (from harvesting permits to the full land title), public management concessions to private forestry firms, or public firms searching for higher profitability.

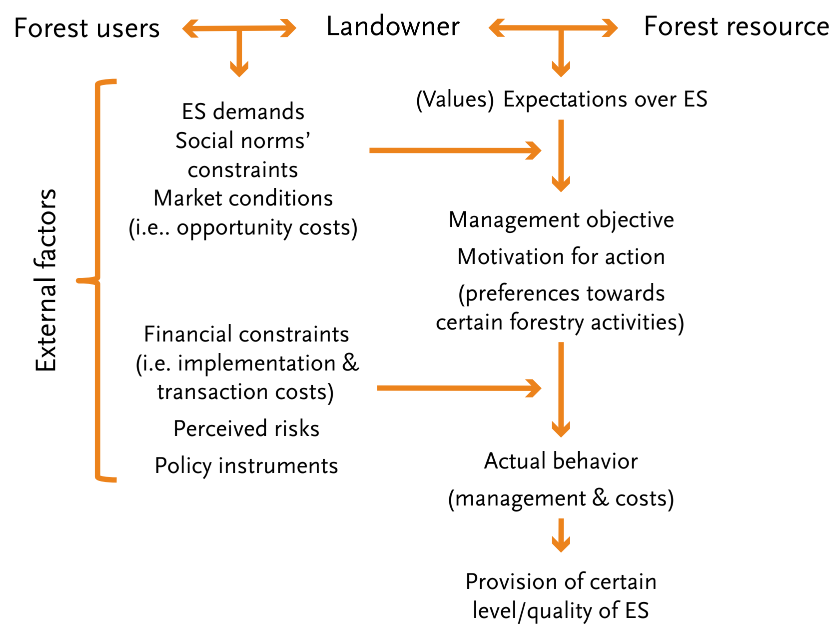

Figure 3. Forest owners' perceptions and motivations in the provision of ecosystem services.

Forest owners derive utility from a variety of sources regarding their forest management (aesthetic values, production, etc.); consequently, their rational preferences would be aligned with management options that increase their utility. The differences in the relationship of the forest owner and her forest resource (e.g. forestry knowledge, forest income dependency, her own beliefs about ecosystem services) may explain some of the variation in objectives. The formation of the forest owner's objective is also affected by external factors, such as market conditions and the expectations on ecosystem services by the society and local communities. Moreover, forest owners are also to a great extent influenced by the culture and social norms in which they are embedded. Depending on the owner's objective for the forest, they elaborate the strategy to pursue it and plan their forestry interventions. Their final implementation depends on both external impediments (e.g. risks, financial constraints, prices) and on their participation in policy instruments targeting forest management. Figure 3 depicts this behavioral model.

Frequently, forest owners with open access properties have experienced an increased recreational use as urbanized populations look for interesting leisure activities; others have learned or been informed about the ecological concerns and arguments for the management restrictions in place in many countries, e.g. when it comes to clear cuts. Forest owners are often aware of the concept that their forests provide important "benefits" for external agents with a meaningful value. However, challenges arise when attempting to implement tools that allow forest owners to capture such values. Given their lack of experience in "marketing" these apparently difficult-to-market-and-control ecosystem, services, owners may remain skeptic about their opportunities for deriving any private benefits. Often a long tradition of open access and non-excludability applies for some of these ecosystem services. Changing the perception of the general public regarding what they can expect to get 'for free' or have to pay for represents a hurdle for the individual forest owner.

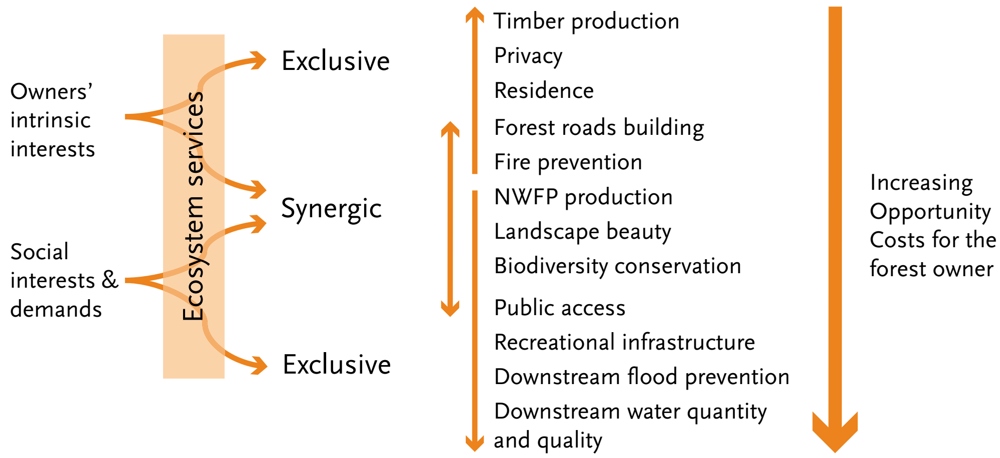

Figure 4. Outcomes from forest management decisions, as expected by society or the forest owner. Some ecosystem services may be exclusive for some agents and others may be shared in certain degree. Landowner's opportunity costs largely depend on his/her intrinsic interest.

It may also be wise not to think of individual forest owners only as individuals, but instead to look also at them collectively, in particular in a spatial context. An agglomeration effect may be desirable if scaling up forest management across the landscape increases effectiveness and cost-efficiency in ecosystem service provision or ensure important thresholds to be reached. For example geographic characteristics and variation of the ecosystem services with scale can cause such situations. However, willingness to cooperate differs across owners. It is more likely that owners with similar motivations gather to face shared concerns, which could be the provision of non-marketed forest goods and services. It is, however, perhaps less likely that such owners with similar objectives are spatially clustered, and hence policy instruments may need to deal with very different forest owner objectives in the same forest landscape.

Appropriate selection of policy instruments should take into account landowner's interests in ecosystem services, jointly with the effects that regulations or monetary incentives can have over them. As long as owners' objectives are aligned with society's interests in terms of ecosystem services it is more likely that they will undertake the efforts to provide them. Synergies may occur with owners' interests: e.g. keeping a suitable habitat for hunting may also provide an adequate environment for certain types of animals and plants. Sometimes forest owners see themselves with a moral obligation to provide ecosystem services free of charge. Figure 4 shows how ecosystem services exclusively benefiting externals usually mean larger opportunity costs for the landowner rather than those where the owner can benefit as well. In cases where the owner's intrinsic motivations for providing ecosystem services do not reach the socially desired level, economic incentives may be necessary to achieve it, to overcome certain constraints (e.g. high initial costs), or to award their behavior (a social recognition as example-to-follow by other owners).

On the other hand, owners' focus may rely exclusively on individually enjoyed benefits and external demands may not be contemplated. Particularly demanding is the case in which socially demanded ES imply a trade-off with the owners' objectives. In these cases, economic incentives may persuade them to change their behavior through increased economic returns, by compensating opportunity costs, additional implementation costs or nuisances in comparison to the management alternatives.

Non-economic policy instruments may also be used, often underpinning economic instruments. Reformulating forest owners' and beneficiaries' rights and duties regarding the provision and use of ecosystem services may coerce these actors to meet social demands e.g. through changes in laws regulating access or harvesting; these regulations are specially adequate when the participation of all forest owners is required or when economic incentives are not able to modulate owners' preferences. Educational tools increasing awareness among forest users and owners, or improving the information on how to approach different silvicultural treatments may modify owners' preferences and therefore facilitate owners' engagement in these activities.

Key messages

- Forest owners should not be considered solely as profit-maximizers, as they typically have a more differentiated set of objectives.
- Forest owners are often aware of and positive to the fact that their forests provide important benefits for society. Capturing part of these values for forest owners income is difficult, but can be achieved with appropriate policies.
- Mapping the variation in forest owners' management objectives will make policy makers better equipped to understand and predict forest owners' responses to policies.
- The better aligned forest owners' objectives are with society's interests in terms of ecosystem services, the more likely it is that they will engage in voluntary instruments to enhance provision.

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3.

Assessing costs of provision

3.1 Costs types and components

Claudio Petucco

Social decision making is based on social costs which include both private costs to e.g. forest owners or other private agents and the wider costs that society could support in any given decision case. Taking this difference into account, this section is devoted to understand cost components and to give some methods to assess the private costs of provision for ecosystem services, using engineering and cost function approaches, as well as the stated cost approach. Where the first two methods attempts to arrive at some 'objective' estimates the latter approach attempts to measure the costs as perceived by the forest owner.

The total cost of providing forest ecosystem services has two main components: on one hand, the direct costs (operational costs and investment costs) which are more easily observable and quantifiable; on the other hand the indirect costs (opportunity costs, transaction costs and feedback costs), which are as well important components of the cost of provision, but usually much harder to identify and measure. All these cost components are commonly assigned to the management actions taken to enhance the provision of the ecosystem services in focus.

The total costs of providing ecosystem services comprises direct costs, easily observable, and indirect costs.

The direct costs are the sum of monetary costs incurred in the specific action implemented for the provision of the public good. For example, the direct costs of enhancing forest recreation may include the cost of creating new paths, the cost of path maintenance, the cost for marking the paths and installing information points or similar. The direct costs include both operational costs and investments costs. The operational costs correspond to the remuneration of the inputs (labour, energy, raw materials, machinery, etc.) used for assuring the continuous provision of the services over time. The majority of the operational costs depend on the effort or intensity of the management actions. However, fixed operational costs are also present; these costs do not vary with the intensity of the management actions, as for instance certain management costs (the cost of inventory assessments and planning for protecting biodiversity or enhancing recreation), as well as administration costs. In contrast, investment costs are are early key actions necessary for the provision of the public good. Normally, investment costs arise in the beginning of the management action, and are necessary to put the action into operation, to create the physical capital necessary for forest environmental services provision. Examples include creation or extension of forest paths, establishing picnic areas, building wildlife corridors, planting or other establishment costs in high risk erosion areas.

The direct costs include all the expenses that can be directly linked to the management action put in place, including both operational costs and investment costs.

In certain circumstances, however, the direct costs are negligible. For instance the direct cost of setting aside forest areas for biodiversity conservation, water protection or carbon sequestration purposes may be implemented by a simple change of the management goal and use for that specific area. This of course does not imply that the cost of providing those ecosystem services is null. As a matter of fact, each management action has also indirect costs, which comprise opportunity costsin the form of income foregone, transaction costs and feedback costs.

The three main indirect costs components are the opportunity costs, the transaction costs and the feedback costs.

In general, the opportunity cost is defined as the forgone benefit that could be obtained from the most profitable, feasible, alternative use of a resource or an asset. In other words, this concept is based on the idea that the cost of using a resource equals the value of what it could be used for instead. As far as forest ecosystem services are concerned, the opportunity cost of a management action enhancing the provision of a particular forest externality could be measured by reference to the returns which could be realised using the land for intensive timber growing or agricultural crops, which are assumed as the most profitable use of the resource. The UN-REDD (United Nations collaborative initiative on Reducing Emissions from Deforestation and forest Degradation) is a significant example of the opportunity cost principle applied to the provision of an ecosystem service. The rationale of this program is actually to offer a monetary compensation for avoiding deforestation in developing countries. Hence, the cost of carbon storage is related to the foregone economic benefits that could have been obtained from timber harvesting and slash and burn agriculture, livestock feeding, soy production or what the suitable alternative land use would be.

The opportunity cost in principle represents the benefits from the most profitable feasible management alternative that the forest manager had to forgo to enhance the provision of a specific ecosystem service.

In many practical applications, however, the business as usual situation is often used as the reference baseline for the estimation of the opportunity cost deriving from an enhanced provision of one or more environmental services. This derives from the fact that forest owners, and in particular non industrial private forest owners, are not necessarily managing the land according to a profit maximisation objective. This implies that the business as usual does not automatically represent the most efficient, profitable, feasible, use of the land resource. This argument can be supported by two main considerations. On one hand, it is generally agreed that the provision of ecosystem services plays an important role in shaping management priorities in private forests, consequently reducing the level of output of the marketable goods (as timber, or non-wood forest product) and favouring non-timber services. On the other hand, there is a lack of information, knowledge, technical means, which act as limiting factors in the efficient management of the forest land. Therefore, evaluating the opportunity cost with reference to the business as usual accounts for these sources of inefficiency, limiting the risk of overestimating this cost component.

Particular caution is required in cases where there is complementarity in the production of forest market goods (i.e. timber, firewood, etc.) and the specific ecosystem service. Complementarity is present when the traditional market goods are produced as by-products generated by the provision of the environmental services or are required as inputs. For example, let's consider an unmanaged forest where a trekking paths network is going to be build. This will require some timber harvest to open the paths and to enhance the scenic view, it may well be that the timber is eventually sold and generates an income. The production of marketable goods as by-products of the environmental good should then be accounted for in the quantification of the opportunity costs. A final important aspect to consider is the spatial scale. The opportunity costs of ecosystem services are not equally distributed over space, and they vary according to the spatial location. In other words, providing a specific environmental service in one forest may have a different opportunity cost than in another forest, due to the different profitability of the respective most profitable, feasible alternatives. Even when the opportunity costs are defined in reference to the business as usual practices; the opportunity costs can vary significantly over space.

Given that the forest ecosystem services often are public goods, transaction costs are particularly important, when the provision is pursued through public policies or coordination activities among forest owners. Transaction costs comprise the resources used to define, establish, maintain and transfer property rights. Generally three main categories are considered according to the transaction time frame: research and information (before transaction), contracting and negotiation (implementation during transaction) and policing (monitoring after transaction). For instance, consider a public administration that stipulates a contract with some forest owners in order to set aside part of their forest to increase the protection of biodiversity. The administration has to gather information on the ecological hotspots, contact the forest owners, estimate an adequate compensation considering their business as usual, prepare the contract, negotiate, monitor the development of biodiversity and control that the forest owners respect the terms of the contract. At the same time, each forest owner has to spend his time to assist at the meetings with the administration, identify the part of the property under discussion, value the compensation offer, and in case of agreement, provide evidences that he/she abides by the rules. Uncertainties about the effort-provision relationship as well as asymmetric information between forest owners and the public agency tend to increase transaction costs. For these reasons, management actions, which are easier to measure and monitor than outcomes, are frequently used as targets when designing policy schemes.

Transaction costs comprise all the expenses linked to research and information, negotiation, and monitoring, in all the cases in which the provision of ecosystem services is implemented through public policies or coordination activities among forest owners.

The third component of the indirect cost is the feedback costs. Feedback costs are the effects that the actions taken to secure the provision of a particular forest ecosystem service may have on other ecosystem services. For instance, an increase in the recreational activities in a forest due to the opening of new trekking paths might disturb the habitat of particular species and eventually cause a reduction of biodiversity protection values. Furthermore, feedback costs comprise the potential negative effects of the increased environmental service provision on neighbouring land uses. A typical example addresses the protection of biodiversity in forest, which may be potentially increase damages on crops in surrounding agricultural lands or in neighbouring forests (i.e. ungulates or wild boars feeding on crops, non-controlled bark beetle infestations, etc.). Feedback costs require knowledge on the links between timber and non-timber benefits, as well as cross-effects between externalities and spatial dynamics. As for the opportunity costs, the relevance of feedback costs depends on the spatial location as well as on the particular management action used to provide the targeted environmental service. Not all management actions have the same feedback costs. In general, assessing feedback costs demand a significant amount of biological and ecological information about the multidimensional impacts of the different management activities on the provision of the ecosystem services. Furthermore, it complicates the assessment of the cost of provision of a given service as the value of feedback effects has also to be assessed to estimate net costs of provision.

Feedback costs refer to the economic impacts that actions to enhance the provision of a specific ecosystem service may have on other ecosystem services or on other land uses in the surroundings.

An important distinction to consider when addressing the cost of provision of forest ecosystem services is the difference between private and social costs. The private costs are typically incurred only by the forest landowners when changing the management in order to enhance the forest ecosystem services provision, though some feedback costs may also be in the form of private costs. The social costs comprise the private costs, but they additionally include all possible direct and indirect costs and expenditures borne by the society. Importantly they do not include any payments or other transfers from society to the forest owners. Such payments are neutral in a social cost sense as they represent an income on the forest owner side. The quantification of the social costs, together with the evaluation of the social benefits, is crucial for the public decision making. However, accounting for the private cost structure is also important for accurately shaping the appropriate policy instruments, in particular for market based instruments.

When quantifying the private cost of provision, it should be clear that forest owners have a personal perception of these costs of provision of forest environmental services. Actually, they often manage their forest in order to satisfy their needs both in terms of wood production and environmental services, as described in an earlier chapter. Consequently, they perceive costs depending on their past experience (if any), on their preference regarding the management of their forest, as well as on their personal characteristics and attitudes. The measurement of the perceived costs provides useful information for the development of tailor made measures of forest policy. Actually, perceived costs also reflect the willingness to participate to the provision scheme. However, the total objective private cost is important to set a baseline reference to estimate the net cost of provision.

In the quantification of costs of provision (both private and public), it is important to consider the fact that forest management is a dynamic resource management problem. In other words, revenues and costs do not arise at the same point in time. Consequently, the different costs components have to be discounted to a reference period via a determined interest rate and only then algebraically summed. Similarly, discounting should be applied to revenues and costs in the determination of the opportunity cost. The discount rate value to be used may vary depending on the cost of capital in the region of interest, the length of the horizon and the risk of the investments. In many practical applications, however, an estimated average bank lending rate may be appropriate. Since the choice of the discount rate significantly affects the total cost of provision, a sensitivity analysis using different values for the discount rate has to be implemented.

It is important to distinguish between private and social costs of provision. The quantification of the former is important for accurately implementing cost efficient policies, e.g. market based instruments. In this latter regard, it is also important to account for the individual perception of private costs and the dynamic structure of the problem.

A summary of the main cost components with some practical examples is presented in Table 2. In general, the quantification of the total social costs, and to some extent the quantification of the private cost structure, is very challenging due to scarce data and a limited amount of options available.In the following sections, the main methods for assessing these costs will be presented as well as some applications from the NEWFOREX case studies.

Table 2. Cost definitions and examples related to the provision of recreational services in a forest massive by a project aimed to build a network of forest paths and recreational facilities for picnics.

Cost type	Definition	Examples	Private-social
Direct costs			
Investment cost	Start-up and capital costs necessary to put the management action into operation	Cost of the project, costs for building the path network, cost of building the parking areas nearby the forest, the picnic areas and installing the information boards.	The investment cost can be either public, private or a public-private combination, depending on who bears the expenditures.
Operational variable cost	Costs related to the remuneration of the inputs (labour, energy, raw materials, machinery, etc.) used for assuring the continuous provision of the services in time according to the effort/intensity of the action	Cost of path maintenance, cost of surveillance, cost of path signs maintenance, cost of waste management, etc.	The operational variable cost can be either public, private or a public-private combination, depending on who bears the expenditures.
Operational fixed cost	Costs related to inputs (e.g., land, labour, energy, raw materials) directly involved in the management action which do not depend on the effort/intensity of the action	Cost of redacting a periodical management plan for the area interested by the recreational activities	The operational fixed cost can be either public, private or a public-private combination, depending on who bears the expenditures.
Indirect costs			
Opportunity cost	Forgone benefit that could derive from the most profitable, feasible, alternative use of the land	Forgone timber income from the land now occupied by the path and the picnic areas.	The opportunity cost is an important element of private cost.
Transaction cost	Cost to define, establish, maintain and transfer property rights	Cost of finding an agreement among the interested forest owners, cost of defining the contracts, cost of monitoring the project	In general, the private transaction costs include only the expenses linked to research, information and negotiation borne by the landowners. The social transaction costs also include the public counterpart's analogous expenses.
Feedback costs	Negative effects of the provision of a particular forest externality both on other externalities and on other land uses	Increased risk of fire, possible disturbances to the local fauna	The feedback costs can be both private and public.

Key messages

- In the quantification of the total cost of providing an ecosystem service it is important to consider all costs components: Direct, indirect, transactions and feedback costs.
- For the provision of ecosystem services, the costs are very often estimated with respect to the management action taken rather than the provided output
- It is important to distinguish between private and public costs of provision. Social costs are relevant for the decision making process, while private costs are particular of interest for policy instrument design and implementation.
- Although the total private cost of provision can be objectively quantified according to these components, forest owners may have a personal perception of these costs.

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3.2

How can we assess costs of provision?

Anne Stenger and Anssi Ahtikoski

As Europe is characterized by large regional differences in the natural conditions, e.g. various climates, institutional contexts, and public demands for forests ecosystem services, there is a need for a large diversity of the methods applied for the cost assessment to reflect this diversity. We generally distinguish between quantitative and qualitative surveys. Quantitative surveys are preferred, but they need an access to basic economic data on forest owner activities, which in turn requires access to a suitably large sample of forest owners that manage their forests and keep accounts. Qualitative surveys can be applied when it is difficult to obtain quantitative data, in particular, when there is a lack of access to a sufficient number of active forest owners. Furthermore, in some countries researchers have experiences that many or even most forest owners rarely carry out management actions and have relative little information about the costs and benefits of different management practices. In this situation asking forest owners to e.g. complete a questionnaire reporting data on management actions and costs is not an appropriate approach to learn about the costs of ecosystem service provision. Nevertheless, quantitative approaches are to be preferred whenever possible, while acknowledging the challenging circumstances in the forest sector where a correct assessment can be difficult for numerous reasons even when active forest owners readily engage in supplying data: Extrapolating results from some forest owners to a large-scale assessment is prone to biases, the typical long time horizon amplifies the uncertainty about the outcomes of management actions, and the typical lags between the change in management and the observed change in ecosystem service provision is a cost in itself that is hard to quantify.

Two main quantitative approaches are considered. Technological approaches focus on outcomes and related objective costs measures, while behavioural approaches focus on the forest owners' decision process and perceived costs. Engineering approaches provide estimated opportunity costs for forest management aimed to enhance ecosystem services, using stand or forest level calculations of loss in discounted net income, net present value. In Chapter 3.4, this methodology is illustrated by discussing case studies where it has been applied to assess the costs of measures aimed at providing enhanced landscape values in Finland and enhanced biodiversity protection in Denmark. The quantitative behavioural approaches discussed focus on revealed cost assessment based on estimation of a cost function and on stated cost assessment based on choice experiments. A cost function approach allows for modelling the joint production of timber and non-timber benefits and the technology could be well represented by a production function or by a transformation function in the multiproduct case, which links produced (private and public) goods and services with the required inputs (labour, land,...).

Stated preference methods are based on a method of asking forest owners for the compensation level they would require for undertaking specific efforts aiming at higher ecosystem service provision. With the current development in stated preference methods, this approach is essentially dominated by the choice modelling approach. The choice modelling method gives the respondent, here the forest owner, the choice between several policy alternatives, for example in the form of environmental contracts, that differs in several ways reflecting the actions to be taken by the forest owners and the compensation she would obtain if she enters the contract, Asking each forest owner to choose several times the preferred contract among several in different sets allow the analyst to estimate the compensation demand for each action included.

Whatever the method, estimating the costs of provision to enhance ecosystem services in most cases will suffer limitations as cost elements like transactions and feedback costs are very hard to assess, also for the forest owners. However, the case studies presented give us a better understanding of the potentials of different methods for cost measurement and confirmed that applying advanced methods for assessing the cost of provision is important for policy making. By combining these estimations with information about the value of enhanced ecosystem service provision, it is possible to estimate a lot better the socially optimal provision of forest externalities. Furthermore, knowledge about forest owners' additional costs associated with an increase in their supply of ecosystem services provides useful information for the design of policy instrument. This includes the development of programs for payment of ecosystem services, cf. chapter 4.

3.3

The engineering cost approach

Erkki Mäntymaa, Anssi Ahtikoski, Suzanne Elizabeth Vedel and Bo Jellesmark Thorsen

An important piece of information for planning and evaluating the viability of an environmental policy targeting ecosystem service provision is an accurate assessment of the costs of the policy. This chapter introduces one method, called the engineering approach that can provide estimated opportunity costs for forest management practices that aim to enhance landscape and recreational benefits. The engineering approach is exemplified below using two different case studies targeting cost estimates from managed forests in Finland and Denmark.

The Finnish as well as the Danish case studies use computational, objective methods with a fundament in stand-level production and management models to estimate the opportunity costs of implementing specific management changes, which enhance ecosystem services from the forest. The opportunity costs are represented by the difference in the net present value (NPV) between the adjusted management schedule enhancing the forest's ecosystem service provision and a conventional management regime (focusing solely on income from timber production). For example, the costs of enhanced ecosystem service provision could be the aggregated value of the reduced revenues the forest owner would experience from e.g. delayed or reduced cuttings relative to the best practice.

The Finnish case study applied a stand-level simulation software to produce alternative forest management practices in the Ruka-Kuusamo area, a major winter sports and nature tourism centre in Northern Finland. The Danish case study focus on the costs of enhancing biodiversity protection in relation to current Natura 2000 policies based on estimates of market prices and state of the art management models and forest growth functions at the stand-level on different site classes in Denmark.

Data for the engineering approach in the Finnish case study

In the Finnish case study, growth predictions were made with the Motti stand simulator which can be applied as a tool to compare stand management alternatives in Finnish conditions. In the case of the Ruka-Kuusamo area the private forest owners' data consisted of traditionally inventoried field data with stand-level forest variables such as basal area, dominant height and stem number assessed. Further, the growing stock was described by tree species and by tree layers. Finally, the measured field data were processed by specific distribution models, which generated the actual tree list to describe growing stock for the Motti stand projections.

Average tree characteristics of the field measurements were 556 stems per hectare, basal area 13.1 m^2/ha, dominant height 14.1 m and average volume 75 m^3/ha. All stands were

Box 4. Predicting forest growth with the Motti stand simulator.

The Motti stand simulator, developed by the Finnish Forest Research Institute (Metla), is a stand-level growth simulator including specific distance-independent tree-level models for predicting such variables as natural regeneration, growth and mortality, as well as effects of management on tree growth. It is designed to simulate stand development under alternative management regimes and in Finnish growth conditions, with specific models for peatland trees.

The Motti stand simulator produces stand projections under various management schedules which are determined by specific user-defined parameters. These parameters include e.g. thinning intensity and timing, stumpage prices, unit costs for silvicultural measures and the discount rate. The performance of the Motti simulator has been assessed in young Scots pine stands and in mixed stands and the reliability of the growth predictions have been evaluated, for instance in intensively managed Scots pine stands. The Motti stand simulator has also been applied at the landscape level.

In general, previous results indicate that the Motti simulator can be applied as a tool to compare stand management alternatives under Finnish conditions. The basic assumptions behind the growth and yield models incorporated into the Motti stand simulator are consistent with the generally accepted principles in empirical modeling of tree growth.

classified as old-growth stands described by biological age, which exceeded 140 years. Most of the stands were spruce-dominated mixed stands with pine and birch admixture (less than 45%), and they represented mesic forests on mineral soils. The stands can be considered to be well-representative of stands in the region with respect to site type as well as to growing stocking. With respect to average stand age in the region the stands in this study were slightly older. The economic data were based on average stumpage prices and silvicultural costs for private forests of the Ruka-Kuusamo area in 2011 produced by the Forest Management Association Kuusamo. For instance, the stumpage price for pine saw logs was 51 €/m^3 at final cut and the cost of a precommercial thinning was 272.2 €/hectare.

Results

In the calculation of opportunity costs for each stand relevant for the landscape alterations, two different scenarios were projected by the Motti stand simulator. First, business-as-usual (BAU) management in which the stand was managed according to the prevailing silvicultural recommendations by the Finnish authorities and second, an adjusted management regime in which the stand was left unmanaged for the next 10 years. In most of the stands, the BAU scenario indicated immediate clear-cutting (in year 0), while the adjusted scenario resulted in growing the stand for the next 10 years, till clear-cutting took place (in year 10). The net present values (NPVs) were calculated for both scenarios, the difference between the NPVs being the discounted income loss associated with taking landscape into account and delaying harvest. The income losses with two different discount rates are presented in Table 3. As can be seen from the results, the income losses were quite reasonable, implying that at least in this case study taking landscape into account in forestry planning, by delaying harvest for 10 years, would not lead to any substantial financial losses.

Table 3. Opportunity costs of enhancing landscape benefits for study stands (income losses due to adjusted management), €/hectare/10 years.

	Discount rate 3%	Discount rate 4%
Average income loss, €/hectare/10 years	27	143
Highest income loss, €/hectare/10 years	129	308

Cost of conservation initiatives related to Natura 2000 policies in Denmark

When dealing with specific management restrictions e.g as those connected with Natura 2000 initiatives, one method could be to investigate the actual costs a forest owner faces if a given restriction is imposed on the property. When the desired forest externalities can be achieved through very specific management changes, it is possible to evaluate the direct costs for the forest owner. Examples of how this may be carried out are shown below for contemporary management changes in productive beech stands.

The cost of a specific management restriction may be estimated based on the change in the net present value of the area before and after the restriction is imposed. Ideally, one would like to assess the change in net present value due to the restriction if the forest is sold, thereby obtaining a market-based estimate of the cost of the restriction. That, as opposed to the method described here could capture any additional costs related to e.g. peoples' reluctance to buy a forest with management restrictions on specific areas. Here, we use the engineering approach and assess the costs of a specific management change by calculating the difference in capital value between two management scenarios, with and without the management restriction.

In Danish beech stands it is possible to carry out a close-to nature regeneration process, which applies reduced soil preparation measures, applies no pesticides and use overstorey trees to ensures that the soil is not bare between two generations of beech stands on the same area; a continuous forest cover model. This management regime is preferred by society for several reasons. Firstly it minimizes the leakage of nutrients to the groundwater and secondly, it preserves the soil microclimate and structure in ways that promote the existence of fungi, insects and through the trophic networks also larger species that depend on these in the forest. Costs can be calculated for different scenarios and site qualities. The expected cost of a management restriction entailing a natural regeneration regime with reduced soil preparation, no pesticides and a prolonged harvest period for the overstorey, are mainly expected to be the longer establishment period for the new generation, possible losses from gaps in the regeneration and the delay in harvesting the overstorey. Examples of cost estimates are shown in Figure 5 below for site quality 1. The figure shows three different cost functions depending on how large the gaps in the regeneration may be due to the management change.

As we would expect, the present value of the losses of course increases with stand age as the actual cash flow differences come closer in time. We note that the effect of age on net present value loss increases much, the more incomplete we fear the new stand to

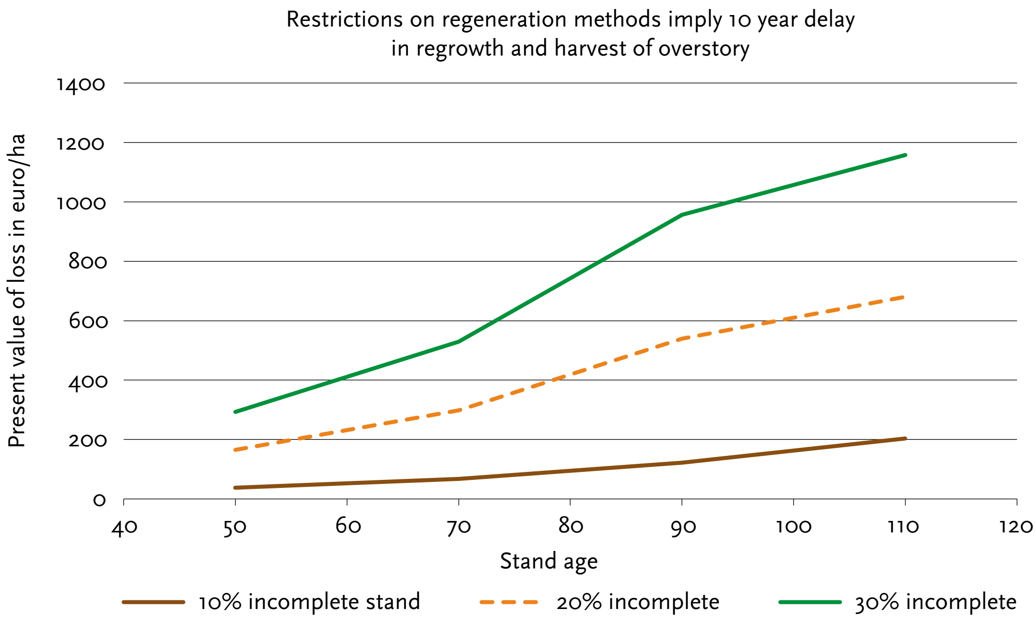

Figure 5. Cost estimates for stands of different age, when restrictions on regeneration methods imply a 10 year delay in regrowth and harvest of overstorey and different levels of incompleteness in the new stand regeneration.

be. The greatest loss of implementing the restriction arises when the age of the beech stand is close to the rotation age (here 110 years), since the additional costs of leaving the overstorey longer and e.g. replant patches will be imminent. For lower site qualities, the restriction in regeneration method implies smaller losses i.e. the loss in NPV for site quality 3 is approximately 40% lower.

Preservation of single trees for decay

The lack of dead wood in forests managed intensively for production is one of the biggest single causes for biodiversity losses in Danish forests. Therefore, one of the initiatives of Natura 2000 seeks to promote that the amount of old growth trees and dead wood increases in forests across the country. A payment scheme targeting exactly this setting aside of single trees has been developed and the opportunity costs of this was estimated using the engineering approach,

When preserving single standing trees in the forest, a similar net present value calculation can be made for each single tree left for old growth and natural decay in the forest. An example of the cost of implementing this for site class 1 in beech is shown in Table 4 for different diameter sizes. From a conservation point of view, it will often have the greatest value to preserve old, large and perhaps already 'uneven' trees – trees which may not have the greatest economic value. In most cases, an estimation of the cost for a preserving a single tree should rely on an individual assessment, however, average costs estimates based on site class and diameter may function as guidelines.

Table 4. Present value of cost (euro/tree) of setting aside single trees for natural aging and decay.

	Site Class 1 Beech:	
Diameter	Quality J/W	Quality C
55	173	201
50	162	184
45	124	141
40	95	108
35	71	80
30	53	60

Key messages

- Engineering type calculations of the forest owners' loss in NPV for given restriction can be used as an 'objective' measure when assessing the cost of enhanced ecosystem service provision.
- The loss is calculated as the difference in the present value of net income between the new management regime enhancing forest ecosystem services and a conventional business-as-usual management regime.
- The engineering approach does not take all economic consequences into account (e.g. reduction in amenity value, potential sales price of property) or forest owners' individual motivations for provision.
- The data for the costs assessment typically come from growth simulations of forest stands or from a mixture of measured field plots and stand projections, combined with existing data on market prices of forest products.

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3.4

An econometric approach to cost of provision

Claudio Petucco

An important method to assess costs of provision of non-market ecosystem services within agriculture is the use of the econometric cost function approach. This approach is able to capture the direct costs and to some extent the feedback costs related to the increased provision of an ecosystem service on the value of traditional marketable products (here e.g. timber and firewood) as well as input uses. This methodology has the advantage of having robust micro-economic theoretical foundations, and the drawback of requiring detailed information and data to be implemented. Compared to the stated cost methods in the following section, this method is based on observing real behaviour and outcomes of landowners' management and from their behaviour the costs are revealed and inferred. Consequently, with this approach it is possible to analyse exclusively the costs of provision of ecosystem services from management actions that have been already implemented, and hence are observable.

The cost function approach aims to assess the direct cost of the provision of ecosystem services based on stated cost information.

Theoretical background

The cost function approach originated from the analysis of the industrial sector in which the firm transforms inputs into market outputs through a production process. Actually, when the firm is maximising its profit, the observed costs represent the smallest expenses needed in order to produce the targeted output (given the actual technology). Consequently, the cost function defines the minimum costs of production for different levels of output. Under sufficiently informed circumstance, the cost function is a "sufficient statistic" since it implicitly incorporates all relevant economic information about the technology adopted. The theoretical economic models proposed in the literature have been accompanied by empirical applications estimating the models' parameters.

The cost function approach has been successfully applied to other sectors of the economy for assessing the opportunity costs of environmental measures. Many applications can be found in the agricultural sector, and few in the forest sector. Under this approach, the farm or the forest property is considered as a production process requiring input (land, labour, capital) to produce one or more outputs. Initially, the econometric analyses of the cost function were limited to marketed goods (crops, milk, beef, construction timber, pulp wood). However, in many cases environmental services are jointly produced with marketed goods. For instance, forests managed for timber production provide certain services as some degree of biodiversity protection, carbon sequestration, water quality and recreation opportunities are still provided. The forest owner is generally not remunerated for provision of these positive externalities, which are often public goods. Hence, it is likely that she is going to supply only a sub-optimal level of these ecosystem services compared to the society needs. To reach the optimal level demanded by the society, they may require the forest owner to modify her behaviour and management and this may possibly increase her costs and lower her profits. Recent studies related to agriculture developed this methodology further in order to accommodate both market goods and ecosystem services as joint productions.

The cost function approach has solid microeconomic foundations and it is derived from the theory of the firm. This methodology has been improved in order to account for the provision of ecosystem services in particular for the agricultural sector.

This methodology for cost assessment has two main advantages. Firstly, once the cost function model is estimated, it is quite straightforward to derive cost elasticities[1] and the single output's marginal costs, which represent the cost of providing an additional "unit" of output. However, it remains to define what "an extra unit" of this output in terms of an ecosystem service means. This depends on the way this service is measured, cf. chapter 2 of the first of these volumes; it could be a unit increase in the biodiversity index, an extra kilometre of path, an additional ton of carbon sequestered, among others.

A second important advantage of the cost function approach is that it can take into account the joint production relationship between marketed goods and non-marketed ecosystem services on the one hand, and between different ecosystem services on the other hand. As already discussed, ecosystem services are often provided together with timber and other market goods. In the economic theory, this multiple output production is called joint production. When we observe a joint production function, it is possible to identify two different categories of goods: competitive goods and complementary goods independently from their private of public nature. "Competitive" sets of goods is observed when increasing the production of one good reduces the provision of the other goods. For example, increasing the extraction of timber from the forest is likely to reduce the level of biodiversity, and its conservation will become more expensive. In contrast, "complementarity" implies that the increased production of one good will result in additional units the other good, whose marginal cost (the cost of providing an additional unit) will therefore be relatively lower. For example, carbon sequestration and biodiversity protection are generally thought to be complementary goods in forest management. The reality, however, is in some cases more complex and the relationship between different goods and services is neither purely complementary nor competitive, and in many cases perhaps poorly understood. That is, the relationship may switch from

1 The cost elasticity is the measurement of how the cost is affected by a change in one explanatory variable (e.g. output, input price, etc.). It is the ration between the percentage change in the cost over the percentage change in the explanatory variable.

complementarity to competitivity as the production intensifies. This information is nevertheless crucial for setting up efficient policy schemes.

Practical applications

In econometrical applications, the cost function is estimated with a regression model in which the total management cost is the variable we want to explain, and cubic meter of timber, the ecosystem services provided as well as the forest's characteristics are the independent variables, also called explanatory variables.

Differently from the agricultural sector, in the forest sector, public databases or accounting statistics of forest owners' economic and environmental performances are generally not available, so it is hard to obtain documented costs and property characteristics. The data used so far has been collected via surveys, and without field measurements to obtain direct measures of the ecosystem services. In these surveys, it is normally possible to monitor only the management action providing specific services (e.g. the forest owner had set aside 5% or the property to protect biodiversity), since measuring the extent of the provided service in the field (e.g. the actual increase in biodiversity) would be a rather costly information to collect at a large scale. Moreover, from the NEWFOREX experience, it was observed that forest owners often did not provide information on their management costs. For instance, only 33% of Danish respondents and 9% of French respondents answered a question about costs of harvesting. These difficulties in gathering the data help to explain the absence of previous studies applying this methodology to forests

The cost function approach aiming to estimate the costs of the joint production of environmental services has not yet been applied in the forestry sector.

Within the NEWFOREX project, a first attempt to estimate a forest management cost function was put in place using the information on the annual gross management costs and the annual timber production provided by 133 Danish forest owners. It was assumed that the forest owners produced two outputs from the forest: timber and biodiversity protection. Given the lack of direct measures of biodiversity protection status, the area of forest set aside for conservation purposes was used as an indirect measure of biodiversity protection. Additional variables were included in the estimated cost function, e.g. the size of the forest, the percentage of broadleaves cover as well as the percentage of non-productive land.

Estimation results[2] suggested that setting aside a certain share for biodiversity protection has a significant positive influence on the gross management cost. Moreover, the estimated coefficient can be directly interpreted as the cost elasticity with respect to biodiversity. It emerged that by increasing the size of the protected area by 1% would

2 The following translog cost function using a second order Taylor approximation was estimated:
$$ln(cost) = b_0 + b_1\, ln(vol) + b_2/2\, ln(vol)^2 + b_3\, ln(biod) + b_4/2\, ln(biod)^2 + b_5\, ln(vol) * ln(biod) + b_6\, ln(X) + \varepsilon$$
where, *cost* represented the annual gross cost, *vol* the annual harvested volume in cubic meters, *biod* the size in hectare of the protected area, and *X* a vector of variable describing the forest property (size, percentage of broadleaves, percentage of non-productive land). All variables were centred at their own sample geometric means (Petucco et al. (2013). Private forest owners' cost function analysis. Manuscript in preparation).

increase the cost by less than 0.21%, suggesting the presence of economies of scale regarding biodiversity protection[3]. A possible interpretation is the presence of operational fixed costs (e.g. defining the borders, changing the management plan) related to setting aside part of the forest property that do not depend on its size. Finally, the estimated model seemed to indicate that harvested volume and biodiversity preservation had a certain degree of complementarity[4]. In other words, the model suggested that increasing the set aside area may end up reducing the (average) marginal cost of producing timber. This could be explained by the fact that the forest owners would likely set aside the less productive part of their property and concentrate the timber production on the more productive part by using it more intensively, thereby reducing the marginal cost of timber production. These results should be treated with some caution, considering the high percentage of non-response in the cost surveys as well as the degree of simplification introduced to cope with the lack of direct observation of the biodiversity level. Moreover, the model does not include the temporal dimension. Hopefully, these limitations will be overcome when better data become available.

Results from a preliminary study on Danish forest owners suggested that this approach can offer insights on the provision of ecosystem services' impact on the cost structure, in particular for direct cost and feedback costs.

Limitations

Although this methodology is very appealing from a theoretical point of view, it is very challenging to empirically implement it in the forestry field. One first problem is related to the length of the production process in forestry. The costs of timber production are not constant in time and are quite high at the beginning of the rotation (i.e. planting, seeding, fencing, pre-commercial thinning, etc.), and then for several years are close to zero until the commercial thinnings and the final felling. Therefore, panel data (surveying the same property over time) should be preferred to cross-sectional data.

A second problem deals with the silvicultural paradigm used by different forest owners and the limitations that in some cases are imposed by the national or regional law. The silvicultural paradigm can be thought as different underlying "production functions" (i.e. monoculture versus mixed forest, even age structure versus multi layers, clear cutting versus single tree selection, etc.). When data are collected, this heterogeneity should be taken into account in order to have unbiased results. In addition, detailed information should be collected on the management operations implemented,

3 The estimated cost elasticity of the size of the protected area (proxy of biodiversity) was obtained by considering the variable at the geometric mean of the sample (volume = 342 m^3, size of protected area = 1.6 ha, size of the forest property = 37 ha, percentage cover of broadleaves = 38%, percentage of non-productive forest land = 19%). The estimated parameter was significant at the 1% significance level.

4 The estimated cross-term coefficient was equal to -0.35 and significant at the 5% significance level.

the related expenses, the prices of inputs used (land, species composition, standing volume, machinery used, labour input, etc.), the types of market outputs produced (construction timber, industrial wood, pulp wood, firewood, etc.) and most importantly direct measures of environmental services (i.e. biodiversity indexes, number of visitors, length of hiking routes, tons of carbon, etc.).

The cost approach relies intensely on data availability and data quality. This is at present limiting its applicability in the forest sector.

Key messages

- The cost function approach is a powerful tool to analyse the cost structure of multi-output forest management.
- It is based on observed or stated costs of specific management activities and hence it can be applied only to existing management actions for enhancing the provision of ecosystem services.
- The cost function approach allows estimating the direct costs of provision as well as the feedback costs, but not transaction and opportunity costs, the latter being often an important component.
- This cost assessment methodology requires and relies on specific and detailed data, limiting its applicability.

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3.5

The stated preference approach to costs of provision

Ville Ovaskainen, Jens Abildtrup, Erkki Mäntymaa, Suzanne Elizabeth Vedel and Bo Jellesmark Thorsen

Why use the stated preference approach?

Apart from the engineering and econometric approach to assessing the cost function, the costs of enhanced provision of ecosystem services can also be estimated using stated preference approaches. Rather than "objective" estimates provided by simulation or revealed cost methods, this method considers the costs of provision as perceived by the forest owner or, more specifically, the minimum compensation that the private forest owner is willing to accept to engage in a voluntary contractual arrangement, which will provide a payment to him against taking actions to enhance ecosystem, services, a so-called PES (payments for environmental services) scheme. Following the stated preference approach common in environmental valuation, a willingness to accept compensation (WTA) measure can be derived through survey-based data in order to anticipate the expected costs of a specified PES initiative.

The basic rationale for the stated cost approach is that the forest owners' compensation claims constitute the supplier's net cost of providing a given level of ecosystem services. The supplier's net costs include the direct, opportunity, and transaction costs accruing to the forest owner minus the owner's private benefit from the ecosystem services provided.

There are several reasons for using the stated cost approach. First, the approach accounts for the fact that the compensation required for a private forest owner to commit to the obligations of a voluntary PES scheme must cover all of the cost components, some of which are not easily estimated and possibly not third-party observable at all.

The simulated opportunity costs – typically forgone revenue due to harvesting restrictions and specific environmentally friendly harvesting and regeneration practices – are by no means the sole, or sometimes even the dominant, cost component. There can be additional direct costs, e.g. related to measures for enhanced recreation opportunities. These are difficult to assess. This is even more so for transaction costs related to information search, planning, and contract making which may sometimes be the most important cost component. Apart from these, the acceptable compensation to the forest owner is likely to depend on his/her own preferences for the ecosystem services being promoted, as well as on his/her preferences for participation per se. The latter may be affected by, for example, a perceived loss of sovereignty in decision making regarding one's own property.

Second, the stated cost approach can be used to assess the costs of new actions for which no data are currently available. Accordingly, stated cost methods may be almost the only feasible way to estimate the supplier's cost *ex ante* for new PES initiatives and action types that are only at the planning stage. While auctions (also known as competitive tendering) can be used at the actual implementation stage of a PES scheme, survey-based estimates of stated costs may be useful and cost-effective in simulating such auctions in advance.

Further, multiattribute methods, such as the choice experiment (CE) method, also allow the assessment of the effects of policy alternatives through the terms of the PES scheme (i.e. contract terms other than the payment). Such survey-based data readily allow us to consider the heterogeneity of forest owners and the related distributional aspects of the policy alternatives.

Despite its obvious merits, the stated cost approach is not without potential problems. One is strategic answering in the sense that forest owners may have incentives to understate their willingness to participate in the different PES scheme alternatives, effectively overstating the true acceptable compensation. A second possible bias, with somewhat contrary effect, is sample selection. In a voluntary survey, it is likely that forest owners who are more interested in the concerned ecosystem service and more favourable to the PES regime may be more likely to participate and respond. Standard procedures for investigating non-response can be used to avoid the potential bias in the average WTA compensation.

Box 5. Cost of provision components that the stated preference approach can reveal related to a PES scheme

- Direct costs include the costs caused by increased and/or changed management actions the forest owner has engaged in a PES contract.
- Opportunity costs comprise of the forgone benefits that could be derived from the most profitable, feasible alternative use of a forest.
- Transaction costs refer to the costs of information search, planning, and contract making related to a PES contract.

Choice experiments for stated cost assessments

In what follows we consider two CE studies that assess forest owners' stated costs of PES initiatives. They share the same basic structure. The respondents were presented with tables (choice situations) each suggesting three alternatives: their current situation and two different contracts. The contracts were presented as varying combinations of the values of selected attributes. These included the management changes required by the contract and the monetary compensation for accepting it, as well as other terms such as duration of the contract. The respondents then selected the alternative they preferred, i.e., the preferred contract alternative or the current situation.

We first consider forest owners' marginal compensation claims related to each proposed management change with other terms of the contract unchanged. These WTA results reflect, directly or indirectly, the marginal supplier's costs with respect to changes in the provision of various environmental services of the forest, such as the landscape and recreational quality. Next we illustrate the assessment of the costs of alternative PES schemes by comparing scenarios that involve a mixture of several management changes.

Table 5. Management changes (contract terms) considered in the Finnish CE study with the estimated marginal compensation claims.

Proposed management changes	Current practice	Alternative practices	Compensation claim, €/ha/yr
Accepting a contract (constant)	No contract		98.6
Harvesting restrictions	Existing previous regulations only	No clear-cutting	–68.0
		No regeneration cuttings	30.0
		No harvesting at all	103.9
Coverage of restrictions	0%	5, 10 or 20% of forest area	5.4[a]
Length of new routes	0 meters	500 meters	n.s.
		1000 meters	n.s.
Contract duration	No contract	5, 10 or 20 years	10.4
Compensation, €/ha/yr	No compensation	30, 60, 120, 180, 240 or 300 €	

[a] Per percentage point; [b] per additional year of contract; n.s.: no significant effect

Enhanced provision of landscape and recreational amenities: A Finnish case

As a means of integrating the interests of tourism entrepreneurs and forest owners, the Landscape and Recreational Values Trading (LRVT) scheme has been proposed in Finland. In this scheme, private forest owners would make voluntary fixed-term contracts whereby they commit to enhance the provision of landscape and recreational values in their forests for a monetary compensation. The contracts would aim to preserve or enhance the landscape characteristics and recreational quality in areas important for recreation and tourism near outdoor recreation routes, shores, and resting places. The funds for the compensations would be collected from tourists as payments connected with the prices of accommodation or other services.

The expected costs are important information regarding the viability of the LRVT scheme. To obtain such information, we use survey data of 471 forest owners in the Ruka-Kuusamo tourism area. The CE approach was applied to assess the minimum compensation that the forest owners would be willing to accept for a LRVT contract. Their compensation claims can be seen as the stated supplier's costs of provision of enhanced landscape and recreational amenities.

The respondents were asked to imagine that a LRVT scheme were to be started in the area. The proposed management changes and other terms of the contracts included the type of harvesting restrictions, their coverage in percent of the property's forest area, length of new outdoor routes, duration of the contract, and the compensation in €/hectare/year. The proposed management changes and the estimated marginal compensation claims are shown in Table 5.

Compensation claims for proposed management changes

To accept a PES contract, the forest owners claim a compensation of 98.6 €/hectare/year irrespective of any specific management changes required. This constant 'threshold value' reflects the forest owner's perceived cost of moving away from the current situation.

A compensation of 30.0 €/hectare/year would be claimed if no regeneration cuttings were allowed, and expectedly a larger one (103.9 €/ha/year) would be required if no harvesting were allowed at all. Considering the coverage of the restrictions, the compensation claim increases by 5.4 €/percentage point/year. Accordingly, the compensation for restrictions affecting 5% of the forest area, for example, should be 27.0 €/ha/year. Interestingly, the figure for prohibiting clear-cuttings is negative (–68.0 €/ha/year). This suggests that rather than claiming a compensation, the forest owners on average considered this change as a benefit and should already be ready to implement it. The establishment of new outdoor routes had no significant effect on the compensation claimed.

The largest compensation claims are associated with changes in the duration of the contract. The claim of 10.4 €/ha per year of contract means that the forest owner's commitment to the enhanced provision of landscape and recreational services for a 20-year rather than 5-year contract would increase the required compensation from 52 to 208 €/hectare/year. This suggests that the forest owners are quite reluctant to accept obligations that restrict their decision-making regarding the management of their property for considerable periods of time.

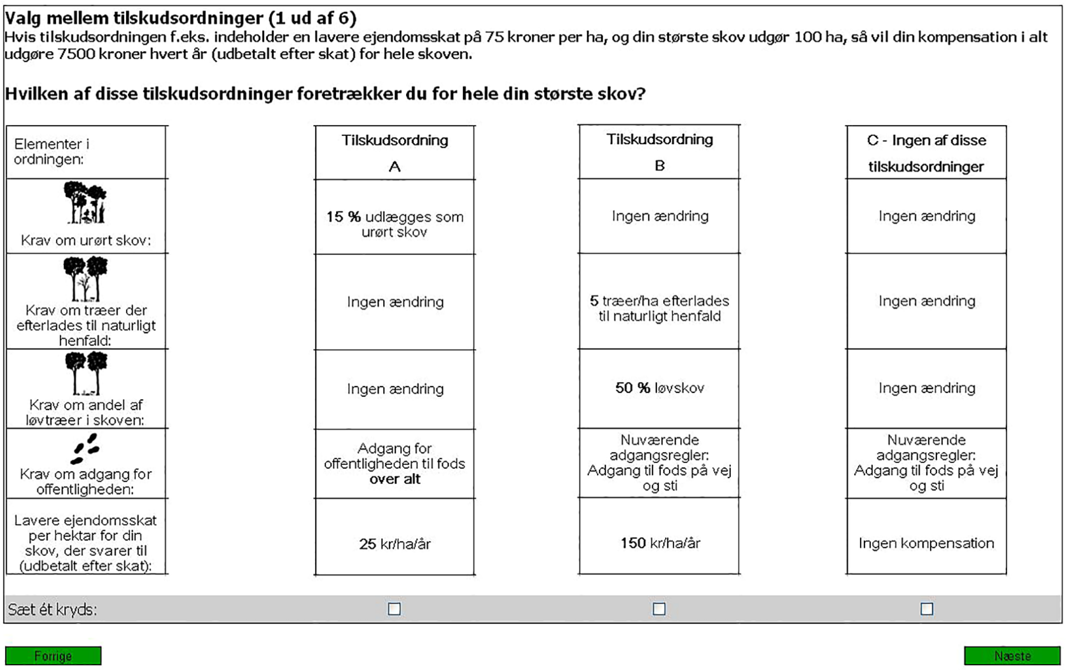
Distributional aspects of policies

The compensation claims for forest owners on average can give a rough idea of the costs of a PES scheme. However, because the stated costs and welfare effects depend on the individual owner's preferences and characteristics, they may vary considerably across different groups of forest owners. The distributional effects are reflected in the way compensation claims vary by the forest owner's income level, for example. In this case, the compensation claims for the 'No harvesting at all' restriction or for a 20-year contract by the high-income group would be around a half of the average level, while the respective compensations required by the low-income group would be approximately three times higher than the average level.

Management changes for biodiversity and recreational access: A Danish case

A similar CE study was made to evaluate Danish forest owners' demand for compensation for specific management changes on their property. The Danish forest area is geographically fragmented and the ownership is distributed across a large number of owners. The implementation of new politically desired changes in management will therefore often involve voluntary schemes targeting a large number of forest owners, each with their view on nature management policies and different management objectives for their land.

A stated cost approach was used to assess forest owners' compensation claims for currently debated management changes related to Natura 2000 policies. The management changes investigated here range from small-scale changes like leaving a number



Valg mellem tilskudsordninger (1 ud af 6)
Hvis tilskudsordningen f.eks. indeholder en lavere ejendomsskat på 75 kroner per ha, og din største skov udgør 100 ha, så vil din kompensation i alt udgøre 7500 kroner hvert år (udbetalt efter skat) for hele skoven.

Hvilken af disse tilskudsordninger foretrækker du for hele din største skov?

Elementer i ordningen:	Tilskudsordning A	Tilskudsordning B	C - Ingen af disse tilskudsordninger
Krav om urørt skov:	**15 %** udlægges som urørt skov	Ingen ændring	Ingen ændring
Krav om træer der efterlades til naturligt henfald:	Ingen ændring	**5** træer/ha efterlades til naturligt henfald	Ingen ændring
Krav om andel af løvtræer i skoven:	Ingen ændring	**50 %** løvskov	Ingen ændring
Krav om adgang for offentligheden:	Adgang for offentligheden til fods **over alt**	Nuværende adgangsregler: Adgang til fods på vej og sti	Nuværende adgangsregler: Adgang til fods på vej og sti
Lavere ejendomsskat per hektar for din skov, der svarer til (udbetalt efter skat):	**25** kr/ha/år	**150** kr/ha/år	Ingen kompensation
Sæt ét kryds:	☐	☐	☐

Forrige   Næste

Figure 6. Visual presentation of the CE to Danish forest owners. The introductory text reads: If the subsidy scheme includes e.g. a lower property tax of 75 DKK per hectare, and your largest forest is 100 hectares, then your compensation will amount to 7.500 DKK every year (in cash after tax) for the whole forest. Which of these subsidy schemes do you prefer for your (whole) largest forest?

of trees per hectare for natural decay to comprehensive changes like setting aside areas as untouched forest, change in tree species from coniferous to broadleaved trees and increased access rights for the general public. The forest owners would select the alternative they preferred from two alternative contracts and the current situation with existing management regulations. Figure 6 shows how the choice question was presented to Danish forest owners in the online web-survey. The proposed management changes with the estimated marginal compensation claims are presented in Table 6.

Compensation claims for proposed management changes

A survey of 283 Danish forest owners was used to estimate how much the owners require in compensation if they were to accept a PES scheme involving the specified management changes. The compensation levels below are per hectare per year payments to the forest owner for the entire forest area – even though some of the initiatives (e.g., setting aside areas as untouched forest) only involve a part of the forest area.

Similar to the Finnish case, the Danish forest owners have a significant compensation claim (43.0 €/ha/year) for accepting a PES contract *per se*. As the compensation for each specific management change is added to this threshold value, the compensation claim for a contract which only entails a 75% broadleaved restriction becomes 43.0 + 7.0 = 50.0 €/ha/year.

On average, the forest owners are most reluctant to accept a PES scheme involving increased access for the general public in their forests. If they are to allow the public access

Table 6. Management changes considered in the Danish CE study with the estimated marginal compensation claims – intermediary results.

Proposed management changes	Current practice	Alternative practices	Compensation claim, €/ha/yr
Accepting a contract (constant)			43.0
Set aside as untouched forest, % of forest area	0%	7%, 15%	0.5[a]
Leave old trees for natural decay	0 trees	5 trees	–1.0
Increase the area with broadleaved trees	0%	Min. 25% broadleaved	n.s.
		Min. 50% broadleaved	n.s.
		Min. 75% broadleaved	7.0
Increase the public's access	Access on roads and paths only	Access on foot up to 15 meters from roads and paths	17.2
		Access on foot everywhere	34.4
Compensation, €/ha/yr		0, 3.5, 7, 10, 13.5, 17, 20.3, 23.6 €	

[a] Per percentage point; n.s.: no significant effect

on foot up to 15 meters from roads and paths, they require 17.2 €/ha/year in compensation, and 34.4 €/ha/year to allow access on foot for the public everywhere on the forest floor. Moreover, 69% of the respondents stated that even if they received an appropriate amount of compensation they would still not be willing to allow access everywhere in the forest. Despite potential strategic answering, this type of survey also shows the scope for which type of ecosystem services the majority of forest owners may be willing to provide through voluntary mechanisms, and for which services only limited results are likely to be achieved through this type of mechanism.

On the other hand, the forest owners on average have a positive attitude towards initiatives to promote biodiversity by leaving old trees for natural decay in the forest and thereby keeping some amount of dead wood. They are willing to accept a smaller compensation when this is a part of the PES scheme. Also, the owners do not require compensation for accepting a restriction of up to 50% minimum broadleaved cover in the forest. For the acceptance of a 75% minimum broadleaved cover, a compensation of 7 €/ha/year is required. Based on these results, a part of the Danish forest owners are remarkably willing to accept high percentages of broadleaved tree species on their property.

If the PES scheme involves setting aside 15% of the forest as untouched, the owner of a 100-hectare forest property requires approximately 750 € in compensation per year. As mentioned above, there may be a bias since this type of survey is likely to attract respondents who are more interested in providing ecosystem services. In the present survey, 60% of forest owners stated that they have already set aside 5% of their forest. The number may suggest that forest owners more prone to take initiatives for biodiversity protection are overrepresented in the sample.

Policy alternatives involving a mixture of management changes

So far, we dealt with marginal compensation claims related to a single management change with other attributes unchanged. However, from a policy point of view it is more illuminating to consider the costs of alternative programmes involving several management

Table 7. Forest owners' overall compensation claims for three different scenarios, €/hectare/year.

Management changes required	Marginal compensation claims	Scenario		
		Least restrictive	Middle	Most restrictive
Accepting a contract (constant)	98.6	98.6	98.6	98.6
No clear-cutting	–68.0	–68.0		
No regeneration cuttings	30.0		30.0	
No harvesting at all	103.9			103.9
5% of forest area	5.4[a]	26.9		
10% of forest area			53.7	
20% of forest area				107.4
5-year contract	10.4	52.1		
10-year contract			104.1	
20-year contract				208.2
Overall claim, €/ha/yr		109.6	286.4	518.1

[a] €/percentage point/year

changes in combination. To illustrate this, three scenarios were constructed for the Finnish case. The overall compensation claims, representing the forest owners' average stated costs for each alternative LRVT scheme, are obtained by adding up the estimated compensation claims for the specific management changes required by the scheme in question.

For the least restrictive scenario with 'No clear-cutting', 5% coverage of the forest area and 5-year contract period, the overall compensation claim becomes 109.6 €/ha/year (Table 6). For the middle scenario, with each of the management requirements tightened by one step in the more demanding direction, the overall claim increases to 286.4 €/ha/year. A further tightening of the contract terms results in the most restrictive scenario with no harvesting at all, 20% coverage of the forest area and a 20-year contract period. The compensation claim then increases to 518.1 €/ha/year, which is more than 4.5 times as high as the compensation for the least restrictive scenario.

How do the stated costs, based on the potential suppliers' own assessments, compare with the simulated opportunity costs of provision of forest amenities? We cannot expect that these be equal or even of the same order of magnitude, as the approaches measure a fundamentally different array of cost components (and benefits, as regards the stated cost approach). In the present Finnish example, the stated costs in Table 7 exceed the simulated opportunity costs for the closest scenario by a factor of ten. This is expected, as the forest owners' acceptable compensation must cover not only the computational opportunity costs (loss of income from harvesting restrictions and specific management practices) and potential direct costs but all perceived costs as well. Maybe the decisive cost category is the transaction costs that are not easily estimated in advance. Further, for the contract to be acceptable to the forest owner the payments should also compensate for the loss of sovereignty regarding the management of one's own property.

Concluding remarks

The above examples illustrate the potential advantages of the stated cost approach in assessing the cost of provision of ecosystem services. Above all, the approach can be used to assess the costs of PES initiatives that are only at the planning stage. The marginal compensation claims related to specific management changes highlight the importance of the detailed terms of the suggested PES scheme. It should be born in mind, though, that the supplier's costs are not an estimate of the full purchaser's budget costs for a PES initiative, as they do not include the scheme manager's transaction and management costs.

It is also worth noting that there is significant heterogeneity in forest owners' preferences for all of the ecosystem services. Part of this is linked to the fact that many forest owners already provide some of these services on their property on a voluntary basis. This is the case especially in Finland, where access for traditional recreational use of the nature is an everyman's right, but also in Denmark with a different recreational tradition. This means that part of the forest owners may accept a contract of the provision of these services without experiencing major additional costs.

All in all, the stated cost approach applied here has the strength of providing compensation estimates of both present and future policies targeting nature conservation and provision of recreational services from forest areas. This is the kind of knowledge that typically would not be available until several years after the implementation of a specific scheme.

Key messages

- The stated cost approach is useful for assessing the total costs as experienced by the forest owner.
- The approach can gather direct, opportunity and transaction costs (for the owner) – adjusted for the potential benefits experienced by the forest owner.
- A significant strength is that new policies or proposed management changes can be evaluated before they are implemented in practice.
- A weakness of the method is the hypothetical setting it relies on. This may induce strategic answering, meaning in this case that landowners might overstate their compensation requirements.
- The supplier's transaction costs, such as the cost of collecting payments for the LRVT scheme in Finland, is not assessed in this approach.

Recommended reading

Mäntymaa, E., Ahtikoski, A., Ovaskainen, V. and Tyrväinen, L. 2013. The Boreal case study. In: Abildtrup J. et al., A report summarizing examples from case studies on the application of cost of provision assessments and the relations to the main findings from the forest owner surveys. Deliverable D3.3 of the NEWFOREX research project. Pp. 73–84.

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4.

Designing economic instruments for ecosystem service provision

4.1 From traditional regulation to economic instruments

Irina Prokofieva and Sven Wunder

Forest ecosystems play a decisive role in the welfare of human societies, but their capability to provide essential goods and services depends upon adequate forest management that ensures their conservation and sustainable use. As a substantial part of forest goods and services are not traded in markets, policy intervention is required to secure these benefits. Policy measures and instruments may take different forms, but they can be broadly divided into three categories: (i) information and environmental education; (ii) command-and-control regulation and public landownership; and (iii) economic and "market-based" instruments (see Table 8).

Information and environmental education instruments include educational campaigns, technical assistance, R&D etc. They are addressed at resolving information-related market failures, which may arise due to the lack of information, information asymmetry, or the lack of skills to use existing information. They are typically used in combination with other categories of instruments.

Command-and-control regulation, also called direct, classic or traditional regulation, relies on the use of state-set rules and penalties to induce a desired behaviour.

The list of instruments may range from state control over the type of activities which are allowed or prohibited (through licences, permits or concessions), prescription of best management practices, determination of quality standards (e.g. water quality) or resource protection (e.g. protection of endangered species or habitats, protected areas). In the extreme case, public intervention may take the form of public ownership and/or management of resources to secure a desired outcome.

Table 8. Types of policy instruments and observations regarding implementation costs. Sources: Pettenella 2013, Prokofieva et al. 2011.

Instrument category	Instrument types	Examples	Direct costs for the public sector	Transaction costs for the public sector	Participation by private actors
Command-and-control regulation	Direct regulation	Prescribed and prohibited activities Zoning & protected areas Licenses/Permits Quality standards Resource protection	Relatively low	Relatively low	Compulsory
	State control	Public (land) ownership Protected areas Public management Public provision	Relatively high	Relatively low	Imposed by the State
Economic instruments	Price-based incentives	Subsidies and grants Tax exemptions and rebates Soft loans	Relatively high	Relatively low	Voluntary
	Tradable permits and offsets	Biodiversity banks Mitigation banking Cap-and-trade schemes	Relatively low	Relatively low	Compulsory for some parties
	Contracts for environmental services	PES and PES-like schemes Public-private contracts	Zero or low for fully private schemes, moderate to high for public schemes	Moderate	Voluntary at least for service providers
Information and education	Eco-labelling and certification	Certification Eco-labels	Zero to low	Zero to low	Voluntary at least for service providers
	Capacity building	Technical assistance Education and training	Moderate to high	Moderate	Voluntary

Economic and “market-based” instruments, also called incentive instruments, operate by encouraging socially desirable behaviour through induced price signal modifications, rather than through prescriptive direct regulation.

Examples of such instruments include different types of subsidies and grants, tax exemption and rebates, tradable permits and offset schemes, certification schemes, contractual mechanisms and market creation. We briefly present each subcategory below.

a) **Subsidies and grants** are by far the most frequent and extensively used mechanisms in forestry all across Europe (e.g. Cubbage et al. 2007; Mavsar et al. 2008). Subsidies are given to support forest holdings economically (e.g. subsidising planting and harvesting of trees, performing timber stand improvement, constructing forest roads), to encourage forest conservation or “multi-functional” or “close-to-nature” forest management aimed at enhancing or protecting biodiversity, soil and water, securing recreational uses of the forest, climate regulation and protection against natural hazards. The agri-environmental measures within the Common Agricultural Policy are good examples of this instrument. Subsidies are also granted for forest extension services, networking support, public relations, or special initiatives (such as Agenda 21 initiatives), marketing measures, establishment of forest owners' co-operation etc. (Mavsar et al. 2008; Prokofieva et al. 2011).

b) **Tax exemptions and rebates** allow differentiation according to environmental impacts, and they also may be used to fund and promote environmentally friendly activities Income tax reductions are typically applied to favour timber investments; or property tax adjustments to benefit forest retention and management (Cubbage et al. 2007). Tax exemptions may also be granted to compensate for legal restrictions on the use of forests, e.g. in protected areas (Mavsar et al. 2008). For example, in Slovakia all protection and special purpose forests are tax free; in France Natura 2000 sites are exempted from tax (ibid).

c) **Tradable permits and offset** schemes are, on the contrary, the least frequently found instruments in forestry in Europe (Mavsar et al. 2008). In trading schemes, an activity is capped at a certain aggregate level (e.g. total amount of diverted water), and permits are allocated and traded among the individuals or firms. An example of such an instrument is the EU Emissions Trading Scheme (ETS), the first and biggest international scheme for the trading of greenhouse gas emission allowances, although currently the forestry sectors is excluded from the scheme. Offsets represent a mechanism by which a loss in ecosystem resources in one area is compensated by a similar gain in another. In an off-set scheme, the landholders or -users creating an undesirable environmental impact in one area are required to offset this impact by investing in conservation elsewhere. The two most prominent offset schemes are mitigation banking for wetland mitigation and conservation banking for biodiversity in operation in the USA, a region where most of the offset schemes are implemented. Similar schemes are emerging in the European countries as well. For example, in France a law requiring avoidance, minimization, and compensation of impacts to the environment (Loi n° 76-629 du 10/07/76 relative à la protection de la nature) resulted in the launch of the first biodiversity bank in 2009 – CDC Biodiversité, a subsidiary of the French financial institution Caisse des Dépôts et Consignations. The biodiversity bank aims to sell biodiversity credits which allow companies to pre-pay, to offset or mitigate the impacts of development (Madsen et al. 2010; Mavsar et al. 2008).

d) **Eco-labelling and certification** aim to provide consumers with information that distinguishes one product from the other (product differentiation). Price premium is paid by the willing consumers that put a higher value on e.g. products meeting certain production process and impact standards. In forestry, two basic approaches to forest certification exist: environmental management systems (e.g. ISO 14001) and forest land management (e.g. Forest Stewardship Council (FSC), Programme for the Endorsement of Forest Certification Schemes (PEFC) and Sustainable Forestry Initiative (SFI) in the US and Canada).

e) **Market creation** refers to the direct exchange of ecosystem goods and services, which requires the existence of a clearly defined and enforceable property right over an ecosystem good or service. For many goods and services markets already exist (e.g. timber, water, hunting and fishing). Yet, for some other goods and services, especially for those with public good characteristics, markets are much more difficult to establish. The difficulties in enforcing property rights, moral considerations or the lack of interest by landowners, are among the major obstacles to market creation for many goods and services (Mavsar et al. 2008). For example, moral objections explain why recreational services in many countries are poorly developed despite the fact that for many recreational uses (e.g. biking, camping, sport events, etc.) the property rights lie with the owner, and no major technical obstacles for their marketing exist.

f) **Contractual approaches for the provision of specific ecosystem goods and services** also belong to the sphere of economic instruments. They may exhibit different degrees of formality and standardisation, ranging from formal contracts sensu stricto to less formal arrangements or agreements between two or more parties. The parties of such contracts can be both private agents (e.g. individuals, companies) and public administration (e.g. government agencies). Budget pressure on government may facilitate the increased involvement of NGOs, international funding bodies, enterprises and other private actors in funding such contracts. The contracts may be oriented at conservation activities (e.g. abandonment of land exploitation or logging, Mature Forest Reserves, see Box 6), at performing specific management practices (e.g. introduction of traditional grazing practices), or at provision of specific ecosystem services (e.g. clean water, recreational opportunities, see Box 7). Payments for environmental services (PES) and different types of PES-like schemes, also usually take the form of voluntary contracts (at least on the side of forest owners).

Box 6. Mature forest reserves (Girona, Spain).

This PES programme, running since 2008 in the Catalonian province of Girona (north-east Spain), aims to promote biodiversity by conserving mature forest stands (stands which have not been actively managed in the last 50–100 years). Forest owners are offered payments for a commitment to leave the stands in natural evolution for 30 years. The programme is funded from the provincial budget and private donors; beneficiaries can be both private landholders and municipalities. The reward they receive is meant to compensate for the loss of profit, calculated using an approved forest management plan.

Box 7. Targeted services provision for specific user groups or for specific goods (Denmark).

In Denmark, there are a few examples of contract-based grants given by public authorities to forest owners to support their adoption of targeted measures to secure specific local public goods or common goods, e.g. allow specific activities in the forest often for limited user groups. The mechanism provides additional recreational opportunities for groups of people where it is believed beneficial to do so from a public point of view, and in areas where their supply from publicly owned forests is insufficient or non-existent. Often these are services provided for a local population only, and many of the activities are directed at children (e.g. exercise track for the public in a municipality, role-play facilities for an association, outdoor education for children). Forest owners in some areas can get additional income from these activities. The buyer of the ecosystem service is often a public institution, a municipality on behalf of the actual end users, but also local associations or NGOs, e.g. within the health and sports voluntary sector.

In the past decades, economic instruments have gained importance as a forest policy tool. Frequently, mechanisms relying on markets allow minimizing transaction costs thus improving efficiency of environmental outcomes. This, however, does not mean that e.g. conservation contracts, biodiversity offsets, payments for environmental services and forest certification are always superior to the traditional regulation. In fact, in many cases the opposite is true.

Policy instrument choice for sustainable provision of forest goods and services requires careful consideration of the socio-economic, environmental and institutional context in which these goods and services are provided, as well as a conscious decision over how much authority the government needs to cede to private actors (forest owners, NGOs, enterprises etc.).

Key messages

- Securing the provision of non-marketed forest goods and services often requires some sort of public intervention – in a form of policy instruments and facilitation – to correct market imperfections and failures.
- Subsidies and tax exemptions have been very popular incentive measures during the last decades.
- Recently, incentive instruments relying on the (re-)definition of property rights have emerged accommodated frequently under the umbrella term "payments for environmental services".
- All instrument types – informational, regulatory and incentive – have their advantages and disadvantages which in many cases are context-dependent.
- As most instruments are seldom used in isolation but are rather in combination with other measures, their performance should preferably be assessed jointly, rather than individually.

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4.2

Shifting roles and powers through voluntary instruments

Davide Pettenella

The still rather limited use of economic instruments in some countries is a clear indicator that economic instruments implementation is negatively conditioned by direct factors connected with the existing systems of market regulation and the perceived and real costs of introducing new tools, as well as by exogenous factors related to the governance system. Among the economic instruments those that are based on voluntary agreements among ecosystem service providers and users are much more exposed to these factors.

Direct factors limiting the use of voluntary economic instruments: In a market where environmental services provision is already conditioned by regulative instruments like constraints (e.g. maximum extension of clear cut areas), minimum thresholds of service provision (e.g. limits to water use) or mandatory practices (treatments against insect attacks or reconstruction works after fire events) there is not much room for introducing voluntary initiatives.

However, when regulative instruments are not associated to any form of direct or indirect compensation, examples of policy failures – like land abandonment or extensivation – are frequent, and voluntary economic instruments can be efficient and concrete alternatives.

In some cases de-regulation can be a useful step for implementing voluntary economic instruments considering that in many cases, for the same ecosystem service, there are several detailed and not always consistent regulations, while there is a lack of clear general rules and regulations (principles and criteria).

On the other hand, de-regulation in the field of ecosystem services provision may rise concerns and opposition by some stakeholders, like environmental NGOs, who tend to oppose the dismantling of traditional systems of environmental protection based on simple constraints and limitations in favor of voluntary economic instruments. The implementation of such instruments is perceived as uncertain, time consuming and not involving all the relevant potential benefit providers. Also ecosystem service providers supported by traditional forms of compensation may contrast the perceived loss of a stable and safe source of income when voluntary economic instruments are introduced. This is a well-know issue in the on-going discussion on the reform of the Single Payment Scheme of the EU Common Agricultural Policy.

Some basic requirements need to be considered to make voluntary economic instruments operational: identification of the clear cause-effect relationships, control of additionality and sometimes also of permanence conditions, leakage avoidance, equity in cost and benefit distribution.

These requirements are associated with the occurrence of transaction costs in stakeholder identification, engagement, negotiation and design of contractual agreements. Occasionally an independent organization is needed to estimate the value of benefits and for monitoring and evaluation the effective outcomes of voluntary economic instruments implementation. All these costs have to be considered in comparing efficiency, effectiveness, transparency and accountability of voluntary economic mechanisms and implementing institutions.

The scale is also a relevant factor influencing the feasibility of voluntary economic instruments schemes. Instruments based on the proximity criteria are normally easier to have implemented and accepted by the relevant stakeholders.

Many successful examples of PES are based on local agreements involving actors that are in direct contact and where ecosystem service beneficiaries can directly enjoy the services and communicate with the providers. Large scale PES are sometimes ineffective and unfair: monetary compensation are difficult to manage at large scales, with buyers not closely in connection with suppliers, relying on intermediaries and on not always very transparent procedures for contract implementation and monetary benefit sharing. Large scale voluntary economic instruments, like those related to carbon offset investments through plantation or deforestation avoidance investments made in far-away countries, have raised quite a lot of criticism connected with the presence of a plethora of intermediaries, high transaction costs and a perceived un-equal distribution of the payments.

Exogenous factors to voluntary economic instruments implementation. Some obstacles to the development of voluntary economic instruments are connected with governance issues associated to the introduction of voluntary economic schemes, in particular of those based on voluntary agreements among ecosystem service providers and users. voluntary economic instruments implementation is reducing the traditional role of public institutions in the use of regulative instruments. Command and control tools need well structured institutions, a top-down hierarchical approach in government and the use of such instruments tends to reinforce the role of the public administration versus the assumption of responsibilities by the producers and civil society organizations. Public institutions may be reluctant when confronted with the need of reducing their traditional role and status. This needed change of focus from the public sector to the market agents is not only a question related to the power relationships among stakeholders: it is a matter of culture and professional background.

Voluntary economic instruments in general, and PES-like instruments in particular, require a different attitude and expertise by supporting agencies: dealing with problems of stakeholder's participation, empowerment of ecosystem service providers, negotiation and conflict resolution. They require skills and know-how that are not always part of the traditional background knowledge of public officials.

In some cases ES are perceived as basic rights (e.g. tap water provision, biodiversity protection) traditionally supplied at zero costs for the beneficiaries. For these services the introduction of voluntary economic instruments may be seen as an unfair form of commoditisation of public goods with the immediate effect of limiting the public access to basic services. Local communities of traditional users mainly in developing countries are in some cases against the idea that basic public services should be transformed in a sort of commodity to be priced and sold in the market. The duties of the State in the provision of these ecosystem services free of charge are emphasized and payment schemes are rejected on ethical grounds.

4.3

Novel contractual approaches and tool design

Sven Wunder and Irina Prokofieva

Within the category of economic incentives, over the past decade or so we have seen an increasing interest in contractual models of ecosystem service provision (through conservation, sustainable forest management, restoration, etc.). Here payments are made directly to forest- and landowners contingent on the environmental desirability of their *de facto* resource management practices. Moreover, in several of these schemes the external beneficiaries of landowners' environmentally benign practices are also directly contributing funds, acting as service buyers rather than delegating the financing function to the state. Finally, some of those incentive approaches are also genuinely "market-based", in the sense that they employ competitive mechanisms to allocate contracts so as to increase cost efficiency of the targeted forest management practices.

It is a full or partial combination of three features – contractual conditionality, the beneficiary pays principle, and the use of competitive forces – which makes for the innovative design features of the forest and land-use based incentives

We describe these in this section. In the practical application of incentive tools, not all of these three elements necessarily go together. For instance, some public environmental contracts have been allocated through conservation auctions that simulate a market on the supply side, e.g. in Australia's Bush Tender programme for habitat protection (Stoneham et al. 2003) and in the US Conservation Reserve Programme for retirement of fragile agricultural lands (Claassen et al. 2008). While the contracts allocated in these auctions have also been conditional, they have not involved user payments, thus lacking the third innovative component.

Arguably the most emblematic type of contractual approaches is payments for environmental services (PES).

The most common definition describes PES as a voluntary, conditional transaction between at least one buyer and one seller over a well-defined environmental service, or a corresponding land-use proxy (Wunder 2005).

Box 8. Private payments for watershed services: Vittel, Nestlé Waters (Eastern France).

Since 1993, mineral water bottler Vittel has been implementing a PES program in its 5,100 ha catchment at the foot of the Vosges Mountains, in order to keep the high quality of aquifer water (Perrot-Maître 2006). The program pays all 27 farmers in the watershed of the "Grande Source" to adopt best practices in dairy farming and land management. It has persuaded farmers to reconvert to extensive low-impact farming, including abandoning agrochemicals, composting animal waste, and reducing animal stocks. The program is fairly complex in design, combining conditional cash payments with technical assistance, reimbursement of incremental agricultural labor costs, and even arrangements to take over lands and provide usufruct rights of the farmland to the farmers. Contracts are long-term (18–30 years), payments are cost-differentiated, and both land uses and water quality are closely monitored over time. Through carefully researched baselines, an improvement of the service vis-à-vis the declining ES baseline is well documented, and the high service value makes the PES investments profitable.

In real-world applications, PES has focused very much on forest ecosystems, and on the provision of four different environmental services (or combinations thereof): carbon storage and sequestration, watershed protection services, biodiversity conservation, and recreational benefits. By design, PES is characterized by quid pro quo conditionality: payments are contingent on monitored contract compliance. The feature that payments could be stopped if the scheme is not working provides an a priori assurance about the effectiveness of PES. Some PES are financed by services users, while in others the public sector substitutes for their role. Only few PES schemes, especially in carbon offset trading, make explicit use of markets; more often than not PES are governed by monopsonic contracts (one buyer or buyer-coordinating body, as compared to a group of sellers).

PES have been applied both in developed and in developing countries. Box 8 describes the Northern Hemisphere private-sector example of Vittel (France) aimed at watershed management for safeguarding drinking water quality; Box 9 presents the case of Bolsa Floresta, an incentive scheme financed jointly by public and private sources, mixing new tools like PES with traditional regulation and integrated conservation approaches.

PES is a tool characterized by voluntary participation – fully so on behalf of landowners as the typical environmental service providers, but sometimes with restrictions for service buyers, especially when services are 'club goods' shared by user groups (such as in many watershed schemes). Simultaneously, to the extent that conditions are being voluntarily negotiated between service users and providers, PES can become a contextually highly adaptive tool, which is thus well-suited to ensure that both parties are genuinely made better off by the intervention. However, as such PES is also potentially exposed to so-called adverse selection bias. For instance, in a PES focused on avoiding deforestation, the landowners who first sign up for the scheme might well be those who would not have deforested in the first place, e.g. because their forest plots were inaccessible and crop marketing costs were excessive, turning PES into a reward rather than a leverage for (additional) change. In both the Vittel and the Bolsa Floresta PES, this was solved by enrolling basically all potential service suppliers in a given area. However, given budget constraints this may not always be possible, posing then the challenge to service buyers to predict where in space payments would have the largest leverage on service supplies – and target interventions to primarily reach those service providers, so that PES can make a real difference.

Box 9. Paying for forest carbon and biodiversity protection: Bolsa Floresta (Amazonas, Brazil).

To counteract incipient deforestation pressures in this remote, underdeveloped part of the Brazilian Amazon region, the State of Amazonas has since 2007 paid households residing in 15 Sustainable Development Reserves (protected areas allowing for smallholder presence) for committing to low-expansive farming strategies that spare primary forests from cutting. Funding sources have included the proper state budget, but also domestic and foreign private carbon-related sources, including one REDD (Reduced Emissions from Deforestation and forest Degradation) project. Typical integrated conservation and development investments (small livestock breeding, forest management, better schools, etc.) have been combined with small, flat PES rates per household. The programme has so far been successful in improving livelihoods and building stronger forest conservation alliances, but will stand up to more comprehensive tests once pressures from a moving agricultural frontier increase (Newton et al. 2012).

Service buyers may also have difficulties in setting adequate payment levels: some service providers may face high opportunity costs (the net losses from deviating from their first-best land use plan so as to provide environmental services); for others these costs may be low or they even gain from environmental service provision.

This problem of asymmetric information about provider costs (providers know their costs well – buyers don't) may offer rent-seeking opportunities to low-cost providers and reduce cost efficiency of the forest-related action. In the Vittel case, payments were carefully customized to each landholder's farming system and opportunity costs – a procedure requiring detailed information and triggering high transaction costs. At the other extreme, Bolsa Floresta employed zero cost differentiation and paid uniform rates, trusting in the relative homogeneity of resident smallholders. As mentioned, inverted auctions – where landowners offer their land for environmental service provision and ask for a certain compensation – can be an alternative, sophisticated tool to reveal disparities in provision cost and in landholders' willingness to accept compensation (Ferraro 2008).

Additionally, potential service providers may also differ significantly in terms of their potential service provision per hectare of (forest) land.

For instance, to protect water quality, keeping areas with high slope and erodible soils under natural forest cover may produce environmental services in avoided damage costs that exceed by orders of magnitude the benefits on a neighbouring flatter plot with more stable soils.

Again, our two PES schemes addressed this challenge quite differently. For Vittel, the intervention and payments size were neatly targeted to the heterogeneity of service provision across the watershed. For Bolsa Floresta, the two targeted service – carbon storage and biodiversity protection – were locally more homogenous in their distribution, depending mostly on tree density for the former and habitat protection for the latter. However, more sophisticated tools may be needed for spatial targeting when various services are being paid for simultaneously. Ultimate, some spatial overlay of service provision levels, leverage/ threat and provision costs may be used to optimize cost efficiency (Wünscher et al. 2008).

Finally, despite their voluntary character, some PES are actually built upon markets emerging around regulatory compulsory requirements, as e.g. in a case of habitat banking that is based on the "no net loss" principle triggering the emergence of compensation and offset mechanisms to mitigate the impact of land development on biodiversity.

Key messages

- Payments for environmental services (PES) are potentially 'smart' (direct, negotiated, adaptive, conditional) conservation tools.
- PES may be financed by service users, or by public sector institutions substituting for those users.
- Adverse selection biases and heterogeneity in service provision or opportunity costs are challenges that need to be overcome by customized design (e.g. spatial targeting and/or differentiated payments).

Recommended reading

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4.4

Examples of novel instruments for enhancing forest ecosystem services

Irina Prokofieva and Sven Wunder

In the past decades, economic instruments of many types have gained importance as forest policy tools. Among them, novel voluntary instruments, e.g. contracts for ecosystem service provision, payments for environmental services etc., attract the most attention. Public mechanisms for ecosystem service provision, that is, mechanisms run and/or financed by public bodies, continue to dominate the policy arena, yet many private or public-private voluntary schemes have also appeared. In this section we present and discuss some of the existing public and private voluntary schemes for four ecosystem services: recreation, water provision, biodiversity, and carbon sequestration.

Recreation

Forests are a preferred environment for recreation and a key component of nature based tourism. The economic importance of nature based tourism is growing faster than any other tourism sector, and it presents a high diversification in activities: forest adventure parks, educational activities, role-play activities, orienteering, forest therapy resorts, etc.

The extent of forest area as well as the access rights for the public to forests for recreational purposes, together determines the opportunities for leisure activities in forests. The issue has received a lot of attention especially in countries with overwhelmingly private forest ownership and limited access rights.

Economic incentives, usually subsidies or contractual arrangements, are used to promote the provision of infrastructure for recreational use on private forest lands.

For example, in Denmark public-private and private-private contracts are used to increase the provision of recreational services, or to allow specific activities in the forest for limited user groups. In Finland, there are some planned experiments, where forest owners are encouraged to take recreational needs and scenic values into account in forest management decisions through a recreational values trading mechanism. In other cases, recreational infrastructure is provided by state forest organizations as part of their duties. These infrastructures are often available free of charge to the users, and thus they do not increase the income of forest owners.

Water

The hydrological services of forests are one of the most important and valuable ecosystem services from forests. Some of these are associated with water quantity and water flows, others are associated with water quality. In some cases water may be the actual service in itself, as in the case of delivering water for consumptive use; in other cases some attribute of water contributes to other services (e.g. providing a recreation environment).

Much of the freshwater in Europe originates in forested watersheds. The biophysical relationships between forests and water are complex and in general are not very well understood. Moreover, they are highly variable from one location to another depending on climate, soils and vegetation types.

Water issues are typically addressed by command-and-control instruments by establishing protective measures (e.g. protection forests) and making use of planning instruments in accordance with related legislation, from zoning plans to, in particular, detailed forest management plans to ensure protective services.

Traditionally, government agencies have made most investments in watershed management, but in recent years a surge in the use of economic instruments for watershed management, especially through PES schemes has been seen.

In principle geographical proximity and the directionality of water as an ecosystem service means that the beneficiaries of improved water quality and/or quantity and the providers of these services by forest management activities can be identified; and therefore, exchange mechanisms can be established between these two groups of involved stakeholders. For example, in Denmark specific contracts are made between water provision companies and landowners aimed to guarantee groundwater quality by providing financial incentives for specific forest management regimes. Similar experiences also exist in other countries.

The emergence of some of these instruments is usually conditional on the existence of formal legal regulations. For example, In Italy, Legge Galli Water Law envisages compensation for water purification services by forests, a mechanism oriented at water quantity and quality and erosion prevention.

Biodiversity

Biodiversity perhaps is the most complex ecosystem service to deal with. Commonly understood as “life in all its forms”, it is a public good, whose definition and therefore measurement are complicated by its multifaceted nature. Existing indicators addressing various dimensions of biodiversity (e.g. species diversity, the amount of deadwood) provide more or less good proxies to measure biodiversity, yet none of them is able to capture all the dimensions of this multifaceted ecosystem service. The spatial and temporal scales of biodiversity are also complex, as biodiversity benefits can be of local or global importance; and some forest activities may take years if not decades to produce an observable impact on biodiversity indicators. All these properties of biodiversity make

Box 10. Novel instruments for enhancing biodiversity.

Land stewardship (LS) activities in Catalonia (Spain): is a bottom-up initiative of environmental NGOs which is being spread quickly during the past decade. The main goal is to engage civil society in the conservation of nature, by actively involving people in volunteering activities that improve ecological characteristics of land sites. LS agreements are done on a voluntary basis between a landowner and an NGO. This instrument has a big component of awareness and communication.

Nature Conservation Agreements scheme (NCA) in Finland, formerly Nature Values Trading (NVT) is a voluntary biodiversity protection tool included in the Forest Biodiversity Programme for Southern Finland (METSO II) 2008–2016, which includes 14 action points. Conservation is based on forest owners' voluntary competitive tendering. Ecological criteria are used to compare and choose the most suitable sites, after which authorities negotiate conservation agreements with the forest owner. Unlike in the former NVT instrument under METSO I programme relying on negotiated payments, in NCA there is no deliberation about the boundaries of the sites and their prices. Compensation is paid based on the costs of nature management on the site and for loss of income. Agreements can be for a fixed term or permanent, depending on a forest owner's preferences.

the development of policy instruments oriented at protecting or conserving biodiversity a challenging task.

Traditionally, biodiversity on the level of ecosystem, species and genes is regulated by means of specific legislation (e.g. nature conservation laws, establishment of protected areas), falling within the domain of command-and-control regulation. Other familiar instruments include different types of payments for access to species or habitat, e.g. hunting, fishing or gathering permits for wildlife species, ecotourism use involving the rights to enter an area, observe wildlife and camp or hike. However, novel economic instruments for biodiversity conservation – such as management contracts for habitat or species' conservation on private land, community concessions in public protected areas, tradable biodiversity credits, biodiversity banks, or labelling of biodiversity-friendly products – are on the rise. Box 10 illustrates two such mechanisms.

Carbon sequestration

Carbon sequestration is one of the best known ecosystem services from forests. Carbon dioxide is withdrawn from the atmosphere by photosynthesis and is sequestered in biomass and soil. Forests play an important role in global carbon cycle, as they are the primary vehicle to remove carbon from the atmosphere. However, forests only sequester carbon while the trees are growing. Therefore, carbon sequestration is a temporary process. Land use and forest management changes have an impact on the age-structure of forests and thereby on the rate at which carbon is being sequestered. Hence, carbon sequestration can be controlled by land-use change to and from forest, and especially by forest management.

Forest conversion is estimated to be the second largest global source of anthropogenic carbon dioxide emissions, responsible for approximately 15–17% of carbon dioxide emissions worldwide. Policies that influence the rate of conversion of forest to other land use, encourage afforestation and reforestation of deforested lands, and avert the loss of standing forests from disease and fire have the potential to have a large impact on carbon sequestration.

Carbon is a global public good, and presumably, carbon sequestration has the same effect regardless of the place in which it occurs. In addition, it is a homogeneous good, a property which is crucial for the creation of any trading or exchange mechanism. The amount of carbon sequestered or emitted from forests can be measured using modelling techniques with sufficient reliability and ease. There are many providers and beneficiaries of this ecosystem service, which again facilitates the application of market-based mechanisms.

Policy instruments oriented at carbon sequestration currently operate at different scales and they include the whole scope of instruments ranging from command-and-control instruments, economic instruments to voluntary actions.

On the international scale, the United Nations Framework Convention on Climate Change (UNFCCC) and the Kyoto protocol within the UNFCCC are the two major agreements addressing climate change and reduction of greenhouse gases (GHGs). The Kyoto Protocol allows for the purchase of carbon credits via forest carbon sequestration as part of the Clean Development Mechanism (CDM). CDM permits a country with an emission-reduction or emission-limitation commitment under the Kyoto Protocol to implement an emission-reduction project in developing countries, earning tradable certified emission reduction (CER) credits, each equivalent to one tonne of CO_2, which can be counted towards meeting Kyoto targets. Since its launch in 2006, CDM has registered more than 1,650 projects and is anticipated to produce CERs amounting to more than 2.9 billion tonnes of CO_2 equivalent in the first commitment period of the Kyoto Protocol, 2008–2012 (UNFCCC 2011).

Carbon markets, both voluntary and compliance, are rather well developed and numerous schemes are operating in this field (e.g. Over the Counter, and Chicago Climate Exchange voluntary markets; EU and New Zealand Emission Trading Schemes (ETS), CDM, Joint Implementation etc. compliance markets), as described in Hamilton et al. (2010). The biggest international scheme for trading GHG allowances – the EU ETS – however, still does not include provisions for covering carbon sequestration from forests, and thus, does not directly benefit forest owners.

On a national scale, economic incentives are typically given to enhance carbon sequestration through afforestation or for bio-energy production. Economic instruments typically include subsidies to afforestation and similar measures, common in many EU countries; however, these are rarely justified on the basis of preserving or maintaining carbon stock. Box 11 presents an example of stewardship payments in Brazil, which explicitly targets carbon emission reductions by means of forest conservation.

On a smaller scale, a voluntary market for carbon has developed among individuals and organisations seeking to become climate neutral in their activities. In Italy, for example, in a framework of the CARBONMARK project, contracts between companies and public forest owners can be signed to offset GHG emissions. Many companies and NGOs set up forest carbon projects especially in developing countries in order to produce carbon offsets, which are later on sold to interested individuals and firms (e.g. the CarbonNeutral company).

Box 11. Bolsa Floresta programme (Brazil).

The Bolsa Floresta (BF) Program employs a set of four incentive-based instruments that are expected to work synergistically in promoting good forest stewardship in the programs target areas, i.e., state-level protected areas with limited sustainable use rights for reserve residents.

The instrument that most closely resembles PES is the cash component of the BF program (BF cash) – a monthly stipend of R$50 (~US$25) conditional on (1) compliance with BF rules, (2) non-expansion of agricultural activities, (3) inform community of fire use for land preparation and use fire breaks, (4) be a member and pay membership fees to the reserve association, (5) send children in school age to school. Non-compliance can be sanctioned after two warnings by suspending monthly payments. The rules thus go marginally beyond pre-existing reserve rules by a) ruling out all new conversion of primary forest and b) requiring agricultural burning to be managed so as to impede large-scale wildfires spreading into the forest. Households must have lived in one of the BF reserves for at least two years to qualify for BF cash.

Lessons learned from the analysis of different economic incentives in the NEWFOREX project

- **Public mechanisms for ecosystem services provision continue to dominate the policy arena.** Among the voluntary mechanisms for enhancing the provision of ecosystem services, the mechanisms run by the public bodies and/or financed from public funds are the most common, the most well known, and the ones that attract the most interest (reflected in the participation rates) by forest owners. This is in line with the results of other studies (e.g. Mavsar et al. 2008, or Milder et al. 2010), which report that publicly-financed mechanisms are the most typical instruments for the ecosystem services provision in Europe. This finding parallels the conclusions from the opinion survey of citizens across Europe, reported in a section below, regarding who should pay for the improved provision of forest externalities, which indicates that the overwhelming majority of the respondents see public bodies as the major legitimate bearers of the additional costs associated with ecosystem services provision. This may well be due to the nature of the studied ecosystem services, which in their majority exhibit the characteristics of public goods (that is, non-rivalry and non-excludability); or may indicate the respondents' overall preferences for government-led initiatives as opposed to private-let initiatives. Further research is needed to shed light on this issue in the context of forest ecosystem services.
- In practice, government bears the costs of ecosystem service provision in at least two instances:
  - When forest land is owned by the government (or other public bodies); and hence it is the public body that is responsible for the provision of ecosystem services; and
  - When forest land is owned by private agents (individuals, groups of individuals, foundations etc.); and the provision of ecosystem services is compensated or (co-) financed from public funds through a variety of incentive mechanisms. The success of these initiatives depends on many factors, such as the institutional environment in which initiatives operate, instrument design, roles and responsibilities of the actors playing a role in the initiatives, as well as the attitudes and opinions of the providers of ecosystem services – private forest owners themselves.

- **Existing policies and organisational structures condition the design and the success of incentive schemes**. From the perspective of institutional design, we find that the design characteristics of the instruments are strongly conditioned by the nature of the actors involved in their development and implementation and by the existing institutional structures. This may lead to certain inefficiencies in design, as in the case of rigid agreement terms in the mature forest reserves programme in Spain (due to the annual budget allocations of the public administration promoting the instrument), or the impossibility to negotiate the terms for the compensation of nature values in METSO II programme in Finland (due to the interpretation of EU regulation prohibiting subsidies to forest values). Conflicts over land allocation among sectors (e.g. agriculture vs. forestry) and the existence of fundamentally conflicting policy goals also hinders the success of some incentive schemes, as in the case of conditional contracts for afforestation in Denmark. This often calls for negotiated and consensus-based solutions with the involvement of all affected parties. In some cases, it may imply that the public bodies need to give up part of their competences to private agents (e.g. those related to the implementation or supervision of the mechanisms), or to engage private agents in funding ecosystem services related initiatives.
- **Forest ownership structure affects the uptake of incentive instruments.** Our study also shows that the knowledge of policy instruments is not spread evenly among private non-industrial forest owners. However, knowing about the existence of incentive schemes does not necessarily increase participation rates in these schemes, which are frequently constrained by the size of the forest land and eligibility requirements. Our findings demonstrate that forest owner associations are key stakeholders for the engagement of individual private forest owners in incentive schemes, as participation in schemes is higher among associated forest owners than among the non-associated ones. Moreover, forest owners participating in one instrument are more likely to participate in other instruments. This is also related to the size of forest property, as we find that large forest owners are more likely to take advantage of different incentive schemes than smaller forest owners. It remains to clarify whether this is simply due to the fact that greater forest land extension allows for assigning different plots of land for different purposes, or whether larger forest owners have a stronger sense of "ownership" of their land than their counterparts, or whether this is related to forest ownership motivations and attitudes.

Key messages

Across the countries included in the study, the clearest tendencies are the following:

- Participation in the MBMs addressing various ecosystem services is higher than in the MBMs addressing a sole ecosystem service.
- Participation is higher in the public MBMs than in the private ones. However, this may be simply due to the fact that there are a larger number of public schemes than the private ones.
- Participation is higher in activity enhancing schemes, than in activity capping schemes. It remains to study whether this is related to the overall activity of the forest owners (active vs. non-active) or other variables.

4.5

Private forest owners' perspectives on policy instruments

Elena Górriz, Florian Schubert and Udo Mantau

There is a wide heterogeneity of owners in terms their management goals and hence the ecosystem services their forests will provide. Aligning forest owners' management decisions with social demands for forest management can be achieved through different policy interventions. From traditional coercion through legislation, governance systems are moving towards persuading owners through informational tools or incentivizing them by means of economic instruments. Within the NEWFOREX project, the factors influencing private owners' participation in policy instruments, especially those of economic character, have been explored across several case studies in Europe. Even if the surveyed populations differ from case to case, as well as the number and type of instruments analysed, several trends across Europe are remarkable.

It has been found that forest owners often have limited information about policy instruments. Yet, being aware of their existence does not necessarily lead to a higher participation rate.

> Forest owners' attitudes towards the goal of the instrument, their opinions on the required commitments, the barriers they may find to enter, or the eligibility criteria can partly explain such (dis-)engagement.

Owners actively managing their forests and those participating already in one instrument are more likely to participate in other schemes. This fact may be related to their easiness to obtain information on the instruments due to their active status within the forest sector and/ora positive experiences in an instrument may encourage them and increase their willingness to enter into a new one. Different socio-economic factors can shed some further light on participation rates (see Box 12).

Membership of some type of forest owner grouping has been found to be a crucial explanatory variable for participation rates.

> Participation rates for different policy instruments are larger among members of forest owners' associations.

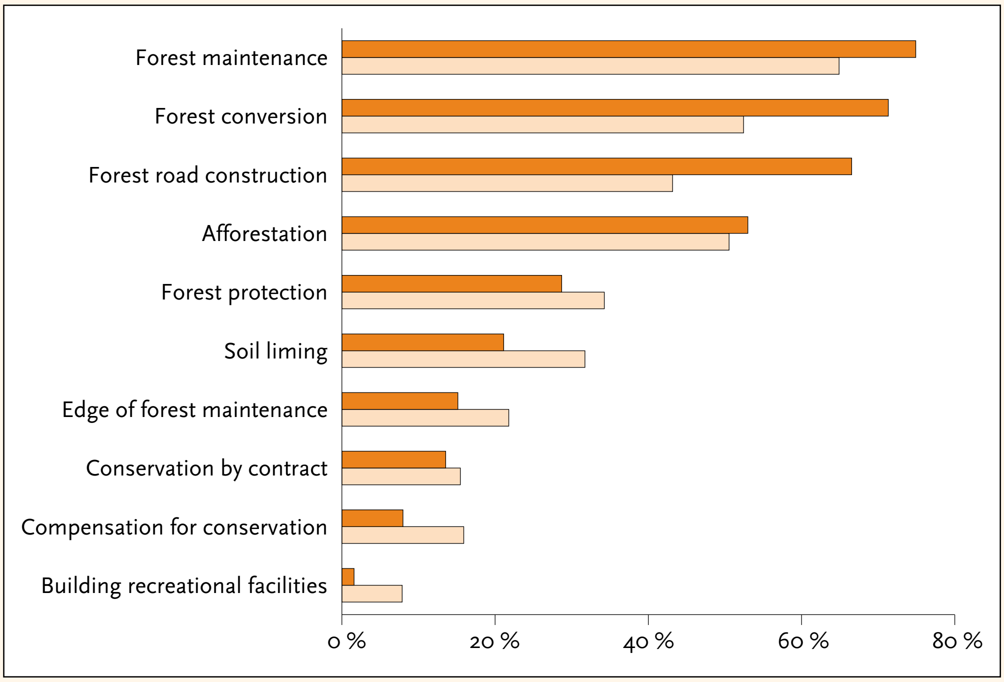
The reasons behind this have been further explored in the German case (see Box 13). The results show that forest owners' organizations represent the most important

Box 12. Influence of socio-economic factors on participation.

For France, Denmark and the Ruka-Kuusamo area in Finland it has been found that large forest owners, those with higher educational levels and better income levels are more likely to engage in different economic instruments targeting ecosystem services. Although these factors can be correlated, this is not always the case. Owning a large piece of land has more probabilities of hosting different ecosystems, and hence having a great variety of management objectives for each plot; additionally, they have a larger margin of maneuver to participate in the instruments only with some portion of the property without restraining their decision-making power in the entire area. More educated owners can be more familiar with bureaucratic processes and information search. Larger incomes give advantage in terms of required cost sharing, risk taking or pre-financing of committed forestry interventions.

Box 13. Influence of being a member of a forest cooperative on participation in subsidies

In Germany, a double survey was launched to forest owner cooperatives across the country: one for managers and another for their members. Results show that the largest group of members is composed by small holders (between 1–5 ha). Half of the queried cooperatives state that they inform their members about available subsidies; in addition, they report their assistance in the bureaucracy to those who participated (over half of the administrative work carried out by the cooperative). Members´ engagement in environmental and production support schemes appears as strongly affected by cooperative's recommendations to participate, as from the large correlation shown in the figure below.



Recommendation rate of cooperatives (darker column) and participation experience of members (lighter column) in various environmental and production support schemes fields (vertical axis). Modified from Schubert, 2013.

intermediaries for participation in environmental schemes between the forest administration and their members, by increasing awareness while also reducing transaction costs and other barriers, as well as harvesting returns to scale.

Forest owner values and the impediments they foresee in cooperating with public agencies or other agemts play a chief role for explaining their engagement in policy instruments. Interviews with forest owners in Catalonia (Spain) have revealed that forest owners' views on other people that use and benefit from their forest are reflected in their preferences for policy interventions (see Box 14). Their participation in voluntary policy instruments would increase if their design includes means to decrease perceived impediments.

Participation rates thrive when the policy instrument is aligned with owners' values

In Germany, members of forestry cooperatives were queried about the relative importance of different values regarding their forest property (see box 15). Consistently, schemes that promote activities with multiple objectives seem to be more attractive than those focused on a single goal. Activity-enhancing instruments tend to be more participated that those limiting the range of actions. These instrument design features (multiple goals, activity-promoter) take advantage of the joint production of ecosystem services, allowing owners to keep on with some of their activities and goals while modifying or adding actions towards an enhanced provision of ecosystem services. Engaging in such smooth tools fits especially with owners with more production-oriented values.

Further aspects of instrument design that are crucial for participation includes contract flexibility, agreement duration, payment timing, control and penalties, the policy mix in which they are embedded or the intermediary. These features have been investigated in the German survey (see box 16).

Policy interventions seeking to modify forest owners' management in a socially acceptable and sustained manner could be tailor-made designed to target the critical actors in view of producing the desired impact.

Typically, policy instruments have included geographical, ecological or socio-demographic requirements to better reach the targeted owners. These characteristics help in prioritizing intervention efforts for the provision of ecosystem services. Results from case studies have shown that taking into account also the preferences of potential participants may improve the performance of the policy instrument. Owners' concerns and interests as well as to constraints they experience to enter into programs or their willingness to cooperate must be uncovered and addressed in policy design in order to introduce means that smooth their engagement, including legal support, credit facilitation or specific awareness approaches.

Box 14. Forest owner preferences for policy instruments and perceived obstacles.

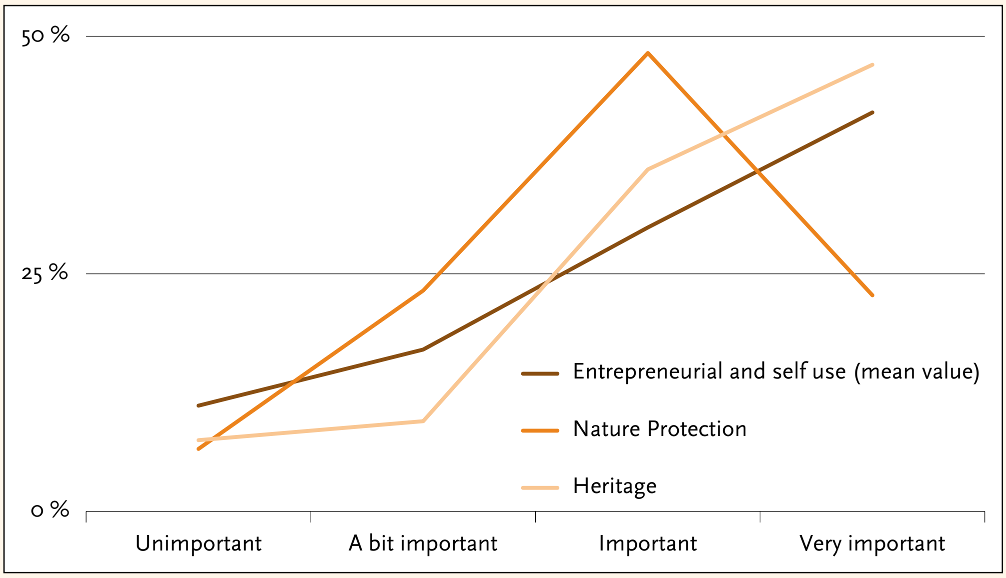
In-depth interviews were conducted to a diversity of forest owner profiles and forest types in Catalonia (Spain). Perceived synergies between owners' management objective and the provision of ecosystem services have been found to increase their willingness to work towards augmenting them. Hence, they are more likely to get involved in any initiative that increases the externality without substantial perceived losses. For example, an owner with rural tourism business may see an interest in increasing bird watching visitors on his property, and therefore may be more eager to participate in a scheme supporting birds' diversity. On the other side, owners highly valuing their privacy may have had negative experiences with forest visitors (e.g. hikers, motorbikes, berry pickers); therefore preferring initiatives that limit and discourage those forest uses. In some cases owners may notice damages and would like to see them restituted by those that caused them. Another category of owners find that their forest property should provide them with monetary revenues; thus, they consider it appropriate to be rewarded for the aspects of their forest management that benefit others. Nevertheless, owners' decision to engage and commit for the provision of ecosystem services depends as well on the perceived presence of obstacles, being of economic nature (e.g. pre-financing, control costs) or social (e.g. local acceptance of new limitations).

Early morning cork oak bark harvesting in a Mediterranean forest. Landowners have shown a general preference towards policy instruments that are compatible with both traditional forestry activities with other ecosystem services provision. Photo: Toni Gorgot.

Box 15. Forest owner values and participation in public subsidy schemes.

In Germany, members of forestry cooperatives were queried about the relative importance of different values regarding their forest property. As illustrated in the figure below, the traditional value of continuing family heritage and forest ownership stands out as the more frequently ranked of higher importance; profit-generating dimensions take the second position, with parallel trends of the entrepreneurial use -understood as the expectations for sales of timber or hunting licenses- and the fulfillment of own requirements -including firewood, hunting, fishing or mushrooms picking for own use-. Responses also show that small owners tend to rank higher the self-use of their forest, whereas larger ones give importance to the entrepreneurial use.

In line with the values-preference rationale, subsidies for afforestation, soil liming and forest road construction are more frequently used in this case by owners that give importance to the entrepreneurial utilization of their forest. Conservation subsidies show a more successful participation rate the more compatible they are with members' values. In this sense, the participation in non-restrictive subsidies, like forest conversion that still allows wood production and covering owner's own requirements (e.g. collecting firewood), is more preferred than participation in restrictive subsides like conservation by contract.



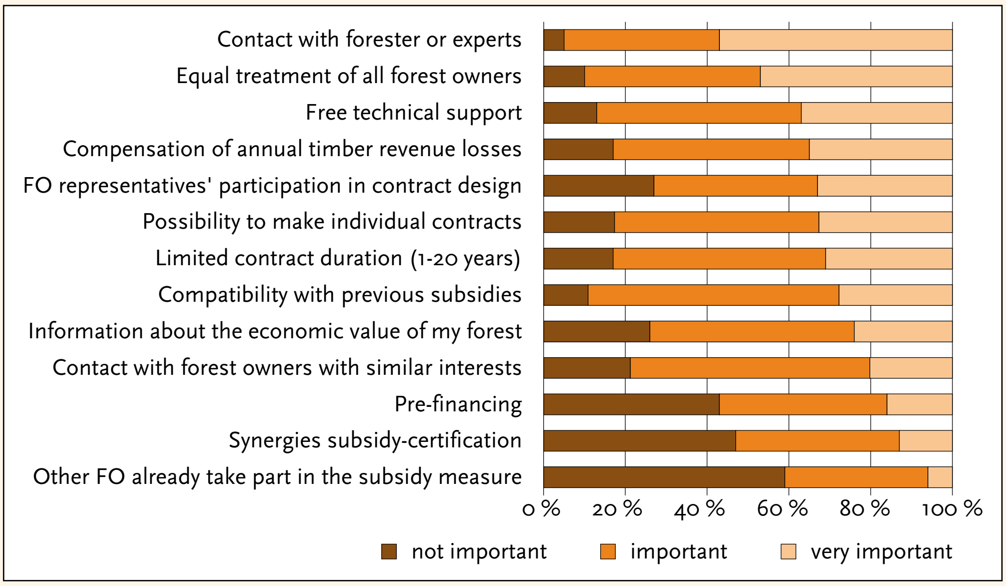
Response rate on the importance of some forest values among members of forest owners cooperatives in Germany. Modified from Schubert, 2013.

Table 9. Average participation rate in different instruments studied according to the instrument characteristics. Data from Germany: Schubert, F. (2013); and from Denmark: Vedel, S.E. (2013).

Instrument features	Denmark	Germany
Multiple ES/single ES objective(s)	7.23% / 6.64%	46.13% / 15.26%
Activity enhancing/activity capping	9.44% / 2.16%	49.85% / 19.06%

Box 16. Forest owner values and participation rates in policy instruments

The figure below shows the relevance of different instrument attributes. Respondents show a higher interest when an option to contact experts or foresters exists; promoting an equal treatment among owners plays also a predominant role. On the contrary, the previous experience of other owners in the subsidy or being helpful for obtaining forest certification are scored as less relevant when making their decisions.



Preferences in subsidy design for members of German forest owner cooperatives. Source: based on Schubert (2013).

Key messages

- Information on targeted owners (frequently difficult in practice).
- Implementation feasibility of the instrument on owners´ forest estate.
- Alignment with values and management orientation of owners.
- Owners' trust in intermediaries and coordination with local authorities and foresters.
- Flexible contracts by authorities and practical support by responsible forest agents.
- Acceptable cost-sharing and work load from the forest owner point of view.
- Habitual participation of owners in policy instruments.
- The more pre-conditions are fulfilled, the higher is the expected participation rate.

Recommended reading

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4.6

Public's view on who should pay for the provision of ecosystem services

Robert Mavsar and Irina Prokofieva

The establishment of economic incentives to enhance the provision of ecosystem services by private forest owners requires, among other, the definition of the optimal level of ecosystem service provision.

The definition of the optimal ecosystem service provision is mainly done by experts.

The definition of the optimal level of ecosystem service provision is typically a very technical task (e.g. establishing relations between the ecosystem characteristics, management measures and ecosystem service provision) and is mainly done by experts (e.g. ecosystem managers, planners). In most cases experts can provide a sound estimate of the potential provision of ecosystem services based on the forests' generating capacity. However, these estimates often do not reflect the beneficiaries' preferences for the provision of the ecosystem services (e.g. types, quantities, qualities, location) nor their willingness to pay for such provision. Bringing the views of the beneficiaries to the picture is important in order to establish the socially optimal provision of ecosystem services, which is based not only on what the nature can produce (eco-physiological supply side) but also on what the society wants in terms of ecosystem services (social demand side). Thus, it is essential to include explicit consultation of the beneficiaries in the process of defining the optimal level of ecosystem service provision. In cases when the ecosystem services are well known (e.g. recreation, drinking water) it is relatively easy for the beneficiaries to provide and express preferences about their desired provision level. However, often we are dealing with ecosystem services, which are very complex and less known or verifiable (e.g. biodiversity protection) by the general public; thus, the beneficiaries might have difficulties to establish clear preferences and express the desired level of provision.

Beneficiaries are expected to be more willing to pay for ecosystem services which they can enjoy directly, such as access to forest recreation.

The second aspect is related to the beneficiaries' willingness to pay for the enhanced provision of ecosystem services. Generally, the willingness to pay depends on the type, quantity/quality of the provided ecosystem service(s) and the beneficiaries' income constraints. When it comes to the type of ecosystem services, we can basically distinguish between ecosystem services that can be directly enjoyed or consumed by individuals (e.g. recreation, mushroom picking, drinking water quality improvement), and those that benefit individuals indirectly (e.g. biodiversity enhancement, carbon sequestration). Ideally, the costs of enhanced provision of ecosystem services should be covered by the beneficiaries of the ecosystem services, regardless of whether these are enjoyed directly or indirectly (so-called "beneficiary pays principle"). However, the beneficiaries might show different perceptions over who should pay and carry the costs for the enhanced provision of these ecosystem services. Assumingly, the beneficiaries would be more willing to pay for those ecosystem services, which directly affect their well being and which require an active use for people to enjoy them as compared to those ecosystem services, which benefit them only indirectly and which all people can enjoy irrespective of their active pursuit of it. In the latter case, the prevailing opinion could be that public administration or even the land owners should bear the costs of providing these ecosystem services. Unfortunately up to date there is very little empirical evidence on whether the opinion of the beneficiaries (e.g. general public) coincides with the "beneficiary pays principle" and if it does not, how does it deviate from it.

In a study that was conducted in five European countries (Denmark, Finland, Italy, Poland and Spain) in the framework of the NEWFOREX project the general public was asked about their opinion over who should pay for the improved provision of different ecosystem services on private lands. The selected ecosystem services included biodiversity protection, carbon sequestration, scenic beauty, water quality and recreation. While the first two services are only indirectly benefiting the respondents and can be enjoyed with no specific action undertaken by the agent, scenic beauty, water quality and recreation provide direct benefits to the beneficiaries, and require agents to undertake some action to enjoy them (e.g. travel or consume). One would expect that respondents would be more willing to bear personally the costs of those services, which they can enjoy directly through an activity they control themselves (recreation, water quality and amenity values), while in the case of biodiversity and carbon sequestration, the enjoyment of which no one can be excluded from, one would expect that people would tend to rely on the public administration (e.g. government) to secure the funds for their provision.

We present the results in Figures 7 through 10. In general, the results demonstrate the tendency of the population to believe that the additional costs of environmental service provision should be covered from public funds, that is, from general tax revenues. This is especially so for ecosystem services with clearly public good characteristics, e.g. biodiversity, carbon sequestration and scenic beauty (see Figures 7, 8 and 9). But even for recreation – which is a clearly user-oriented service, the majority of the respondents (except for the Spanish and Finnish respondents) favour government pay option over the other alternatives (see Figure 10). Paradoxically, Spanish respondents think it should be the obligation of private forest owners to bear the costs of additional recreational services, whereas in Finland the respondents tend to accept that these costs should be imposed on forest users – visitors. Finnish respondents are also the most reluctant to impose such costs on forest owners.

While the overwhelming majority does support the government pay option for almost all studied ecosystem services, we have observed slight differences in opinions that may

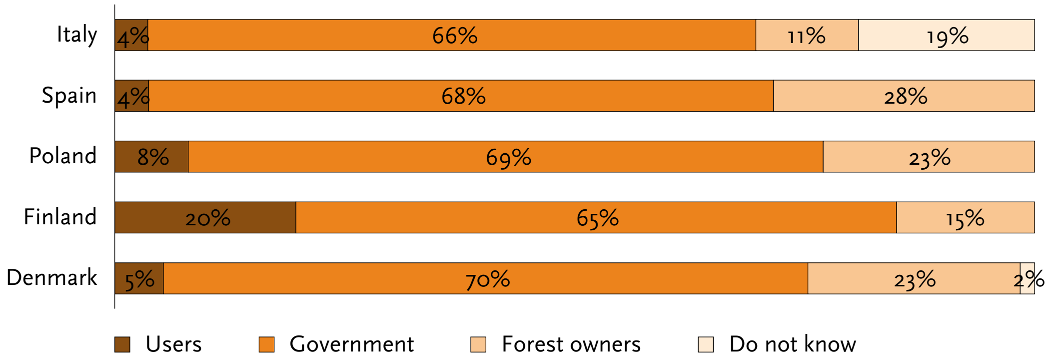

Figure 7. Who should pay for biodiversity improvements: comparison across cases.

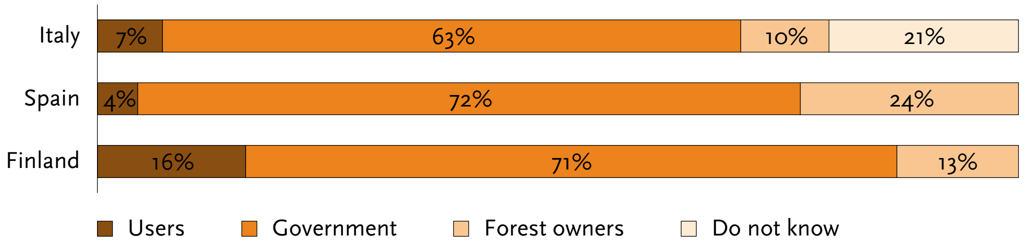

Figure 8. Who should pay for increasing carbon sequestration in forests: comparison across cases.

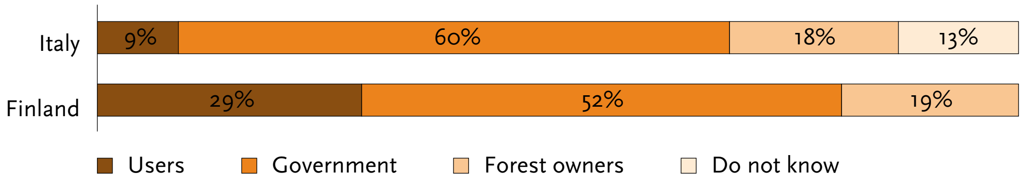

Figure 9. Who should pay for enhancing scenic beauty: comparison across cases.

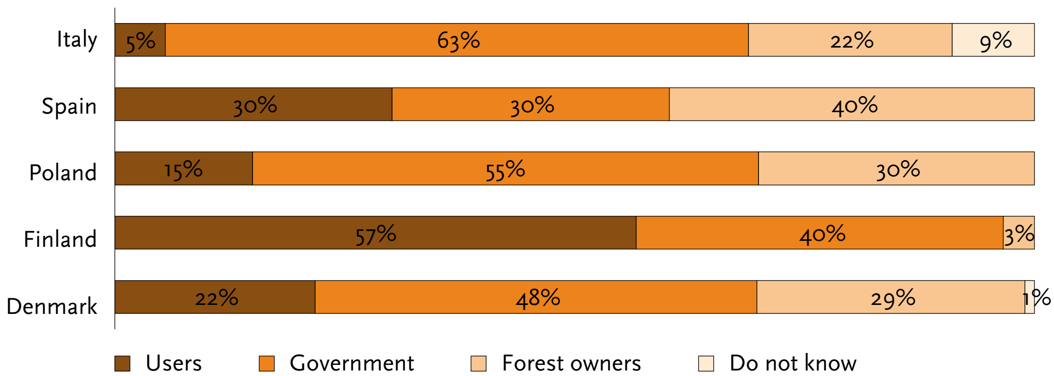

Figure 10. Who should pay for improvements in recreational services: comparison across cases.

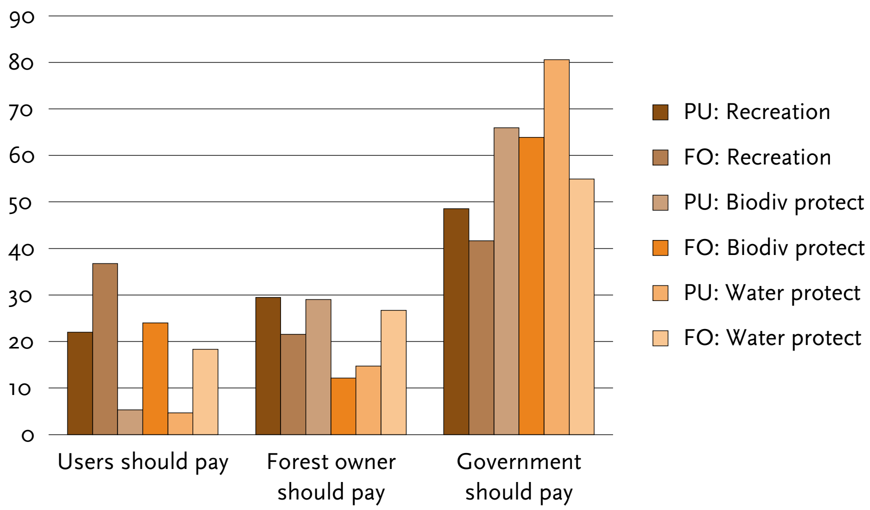

Figure 11. The comparison of public and forest owners opinion on who should pay for the improved provision of several ecosystem services in the Danish case.

be worthwhile to keep in mind especially when designing public support campaigns for certain types of incentive schemes. These differences, can be observed for different geographical (Nordic vs. Southern countries) and demographical (e.g. older vs. younger, low income vs. high income, lower educated vs. higher educated respondents) segments of the population.

Different stakeholders might express diverse preferences about who should bear the costs for the optimised provision of ecosystem services.

In one of the countries, namely in Denmark, a similar question on who should pay for enhancing ecosystem services was also asked to forest owners. Figure 11 presents the results of both public opinion (PU) and forest owners opinion (FO) on this subject. As the results illustrate, while the opinions of general public and forest owners are quite aligned, the latter ones seem to favour slightly more the users pay option, even for services such as biodiversity and water protection. Surprisingly, though, among forest owners we also find a higher percentage of those who think that forest owners (and not the government) shall carry the costs for for improved water protection.

These, empirical examples clearly show how important it is to explore the opinion about payment alternatives among involved parties. Unfortunately, this step is often omitted, due to time or financial constrains. Nevertheless, it is essential as it gives information, which can significantly contribute to a more appropriate design of the instrument or point out, which preparatory actions (e.g., education and information campaigns) would be needed before implementing it. Once again, we would like to point-out that the main advantage of economic instruments, when compared to more traditional approaches, is their case specific design, which should enable a superior efficiency and effectives, but also the acceptability by the involved parties.

Key messages

- In theory we would expect that beneficiaries are be more willing to pay for the provision of ecosystem services, which they can enjoy directly (e.g., recreation, landscape aesthetics), and less for indirect benefits (e.g., carbon sequestration, biodiversity enhancement).
- There are only few empirical studies about who should pay for the optimized provision of ecosystem services, but the results show that beneficiaries prefer that the government would bear the additional provision cost.
- The understanding of stakeholders preferences is important for a more efficient and effective implementation of economic instruments.

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EUROPEAN FOREST INSTITUTE

Yliopistokatu 6, FI-80100 Joensuu, Finland
Tel. +358 10 773 4300, Fax. +358 10 773 4377
www.efi.int

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