Analysis of Different Approaches and Methodologies on Valuation and Payments for Forest Ecosystem Services in the Pan-European Region

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Preface

Forest ecosystems provide a range of goods and services from traditional natural resources such as timber to clean water and air, protection from floods and erosion, as well as recreational services including cultural values. Forest ecosystem services, and the natural capital stocks that produce them, make significant direct and indirect contributions to national economies and are of fundamental prominence in human health and well-being.

Although the importance of ecosystems to human society has ecological, socio-cultural and economic dimensions, expressing the value of forest ecosystem services in monetary units is an important tool to raise awareness and convey the importance of ecosystems to policy makers, to mention just a few application areas.

The valuation of ecosystem services and their integration into policy and decision making practices has been matter of debate ever since the concept first emerged in the early 1990s.

The issue of forest ecosystem services, their values and payments for them has been reflected also in several policy documents of the FOREST EUROPE process.

At the Seventh FOREST EUROPE Ministerial Conference in Madrid in 2015, the ministers responsible for forests in Europe adopted the Madrid Ministerial Resolution 1 “Forest Sector in the Center of Green Economy” where they committed themselves inter alia to incorporate the value of forest ecosystem services in a green economy. Besides the Madrid Resolution, also other commitments related to forest ecosystem services were adopted by previous Ministerial Conferences on the Protection of Forests in Europe. Essential benefits provided by forests and the importance of the value of forest services had already been recognised in Vienna Living Forest Summit Declaration “European Forests – Common Benefits, Shared responsibilities” (2003), and Oslo Decision “European Forests 2020” (2011). Water related services were addressed in Warsaw Declaration (2007) and Warsaw Resolution 2 “Forest and Water” (2007).

Following the Madrid Ministerial Conference, the Expert Level Meeting held on 11-12 May 2016 in Bratislava, Slovakia approved the FOREST EUROPE Work Programme for the period 2016-2020. In line with the Work Programme, FOREST EUROPE continued exploring different approaches to valuation and payments for forest ecosystem services existing within the pan-European region in order to identify possible methodologies and replicable experience.

To fulfil this activity an Expert Group on Valuation and Payments for Forest Ecosystem Services was established for the period 2016-2018. The Expert Group dealt with three specific tasks:

1. Analysis of different approaches and methodologies on valuation and payments for forest ecosystem services,
2. Collection of case studies of valuation methods and payment mechanisms implemented in signatory countries,

Based on the outcomes of these tasks, a web-based portal was established to serve as a platform for knowledge and information exchange on valuation methodologies and payments for FES as well as sharing best practices in this field.

The aim of this publication is to review the state-of-the-art in valuation and payments for forest ecosystem services in the pan-European region and create a foundation for policy recommendations and conclusions on the subject matter.
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Executive Summary

Forest ecosystems provide a multiplicity of benefits of crucial ecological, social and economic importance for the sustainability of our society. These benefits are known as ecosystem services. These include, for example, the provision of food, clean water, timber and fibre (provisioning services); the regulation of air quality, climate and flood risk (regulating services); opportunities for recreation, tourism and cultural development (cultural services).

The importance of sustainable management of forests and the services they provide has increased during years. This is also reflected in a number of policy commitments adopted by ministers responsible for forests of FOREST EUROPE signatories as well as EU policy documents, such as the new EU Forest Strategy, the EU’s Bioeconomy Strategy and its Action Plan and other related documents.

The Madrid Ministerial Resolution 1 “Forest sector in the center of a green economy” (2015) recognized the key role of forest ecosystem services in the contribution of forests to a green economy. Prior to the Madrid Resolution, essential benefits provided by forests and the importance of the value of forest goods and services had already been recognised in Vienna Living Forest Summit Declaration “European Forests – Common Benefits, Shared responsibilities” (2003), and Oslo Decision “European Forests 2020” (2011). Water-related services were addressed in the Warsaw Declaration (2007) and Warsaw Resolution 2 “Forest and Water” (2007).

This study provides a review of the state-of-knowledge regarding the economic valuation and financial mechanisms for forest ecosystem services in order to provide background for policies and practice.

This publication is in line with the Action 4.4 “Incorporating the value of forest ecosystem services in a green economy” of the FOREST EUROPE Work Programme 2016–2020.

In this context, the FOREST EUROPE Expert Group was established to continue analysing different economic valuation methods as well as different financial mechanisms for forest ecosystem services implemented in the pan-European region.

There is a variety of methods and approaches to determine the value of ecosystem services. Economic valuations usually attempt to elicit individual preferences within the general public for changes in the provision of services or in the state of the environment in monetary terms. These are based on the fundamental principles of welfare economics; whereby the changes in the well-being of individuals are reflected in their willingness to pay or willingness to accept compensation for changes in their level of using a particular service or bundle of services.

Some of the respective methods rely on actually observed behaviour of the users for these services (i.e. revealed preference approaches). However, the applicability of these methods is limited only to a few forest ecosystem services (e.g. recreation, tourism and amenities). Other methods are based on hypothetical rather than actual behaviour data and use surveys and directly or indirectly ask users about their willingness to pay for certain services (i.e. stated preference approaches). The methods in this group can be applied to all types of forest ecosystem services and allow to estimate both use and non-use values. Anyhow, their main disadvantages are that they are based on hypothetical situations (no real market transaction is performed and the received answer might not reflect the real situation).

Based on the review of valuation methods it can be stated that all valuation techniques have their advantages but show also methodological
limitations, hence it is necessary to have validated and accepted approaches for valuing different groups of ecosystem services. This is because some ecosystem services require specific suitable methods, and one size doesn't fit all in this case.

When deciding which method to apply, the general recommendation is, first, that it must fit the given valuation problem; and second, to take into consideration complexity of the method and resources required for its implementation (in terms of data, time, analysis skills, and other requirements). However, the final selection of a valuation method is determined by a number of other factors and conditions such as the services to be valued and their context, geographical scope (local, regional, national, international), data availability, available time, financial resources, and experience.

In the last decades, valuation methods and approaches have reached a considerable degree of sophistication. Despite this, some elements still exist to be considered when selecting the valuation methods, such as:

- Interdependence of ecosystems and their services – for valuation, this means that economic value of any one service may depend on its relationship with other services;
- Marginality - economic valuation is meaningful when considering marginal changes in the provision of ecosystem services;
- Double counting - some ecosystem services are not complementary, the provision of one is precluded or negatively impacted by others (trade-offs);
- Spatial issues – valuation should take into account the complete population affected, whose values may be affected by the changes in ecosystem services supply. To estimate appropriate values, it is necessary to understand whether an ecosystem service is impacting at local, regional or global level.
- Temporal distribution of costs and benefits. This is normally done by discounting, using an appropriate discount rate, which converts all costs and benefits to present values so that they can be compared;
- The state of ecosystems conditions.

Services that ecosystems provide depend not only on the scale and function of the ecosystem but also, crucially, on its conditions.

- Dealing with uncertainty surrounding both the knowledge regarding functional aspects of ecosystems and valuation of ecosystem services;
- Value transfer and knowledge gaps. Issues are related to the need for good quality studies of similar situation. Usually, databases of valuation studies contain rather general information and do not cover methodological details and/or are not generally accessible, which is not sufficient for value transfer.

Economic valuation can be an important step for developing mechanisms to measure the benefits of the services and in designing finance/incentive systems such as payments for ecosystem services (PES). PES covers a variety of financing arrangements through which the beneficiaries of ecosystem services pay the provider of those services, thus offering incentives for protecting or restoring of ecosystems and thus maintaining or increasing the supply of such services.

A PES scheme often involves different actors. The latter term encompasses anyone with a “stake” in an issue, both those actors with influence (i.e. those controlling a resource), and those influenced (i.e. those affected by a change in the resource).

In practice, PES often involves a series of payments to service providers i.e. land or other natural resource owners or managers in return for a guaranteed flow of ecosystem services or, more commonly, for management actions likely to enhance their provision over-and-above what would otherwise be provided in the absence of payment. Payments are made by the beneficiaries of the services in question, for example, individuals, communities, businesses or government acting on behalf of various parties.

Although PES theory mainly refers to these two actors, other actors such as knowledge providers, intermediaries and/or donors can influence the design and implementation of the contractual agreement.

Main PES classifications are based on the type of actors such as public, private, and mixed public-
Private. Public and private schemes may adopt different financial arrangements regarding the compensation to sellers and the collection of buyers’ contributions. The most common financial arrangements include subsidies, tax-based contributions, direct compensation, land lease, eco-sponsoring, donations and certification.

For a PES scheme to work it must represent a win-win situation for both buyers and sellers because otherwise voluntarily participating actors would not want to engage. PES may be positive from a buyer's perspective if the payments are less than those associated with any alternative means of securing the desired service, or with the value of the otherwise lost environmental service. PES schemes may be positive from a seller's perspective if the level of payment received at least covers the value of any returns foregone as a result of implementing the agreed interventions (however, there are also transaction costs that need to be covered as a minimum).

PES systems work best when services are visible or quantifiable and beneficiaries are well organized (e.g. in associations), and when land use communities are well structured, have clear and secure property rights, and strong legal frameworks. These conditions minimize sources of interference with the newly created market and reduce transaction costs.

PES should be also seen as a policy tool with several advantages and opportunities. One of the foreseeable advantages of the successful implementation of PES schemes is to maintain a sustainable supply of non-market forest services. PES can create incentives for the providers of forest services to manage forests following a multifunctional approach and for keeping a constant or increasing supply of services without any loss. The buyer of ecosystem services supports the ecosystem services provider by payment, which has to compensate for any reduction in timber production.

PES schemes are often recommended as being more flexible, more easily applied and more cost-effective, allowing high customization to local circumstances.
PES instruments, because of their voluntary nature, offer a less prescriptive and coercive approach and therefore may be a more feasible instrument in practice in some situations, especially it seems to be most effective in private PES schemes. Voluntariness provides flexibility in decision making. The voluntary nature of PES gives the opportunity to negotiate deal details between stakeholders without any restrictions and limitations.

Although PES programs are not designed for wealth redistribution, there can be important synergies with social aims. This might specifically support the European policies for rural areas.

PES schemes may also provide non-monetary benefits such as training, specialist advisors, infrastructure improvements or technical support. Rural communities can benefit from increased knowledge of sustainable resource use practices that are usually connected to PES through the provision of training and technical assistance.

On the other hand, various difficulties and challenges exist in the implementation of these financing mechanisms. The most common challenges are associated with the definition, understanding, measurement and economic assessment of ecosystem services at appropriate scale and precision.

In addition, the application of a specific payment scheme depends on the interest and willingness of involved actors, laws and regulations in place and sufficient financial resources.

High transaction costs including the cost of identifying and selecting service providers, attracting potential demand/buyers, negotiating and developing contracts, training, monitoring, reporting and follow-up activities, etc. also remain constraints in applying PES.

Another precondition for PES implementation is the secured ownership and tenure rights of forest land. If property or use rights are unclear, the buyer of the service cannot define the conditions of payment.
In the Madrid Ministerial Resolution 1 “Forest sector in the center of a green economy”, ministers expressed their commitment inter alia to “Promote the exchange of information on methodologies and practices on the valuation of and payments for forest ecosystem services as well as policy approaches to this end”. In line with this commitment a web-based portal for forest ecosystem services was established to serve as a platform for knowledge and information exchange on valuation methodologies and payments for FES as well as sharing case examples in this field.

The web portal is a tool that aggregates published information and case examples of valuation and payments for FES in one place via a simple browsing system. The portal comprises three modules: (i) Introduction to forest ecosystem services; (ii) Valuation of forest ecosystem services; and (iii) Payments for forest ecosystem services.

The modules include interactive tools such as schemes and map with case examples collected from FOREST EUROPE signatory countries. Interactive schemes provide simple orientation within different valuation methods and access to their examples and thus facilitate decision-making on FES and select appropriate method for valuation of particular FES.

An important part of the portal is an interactive map of FOREST EUROPE signatories that present an overview of financial mechanisms for forest ecosystem services implemented (or to be implemented) in the pan-European region. There are examples of well-functioning and/or developing PES schemes in the forest sector in FOREST EUROPE signatory countries. These examples represent a basis for development of new PES. However, identification and assessment of local socio-ecological conditions needs to be taken into account.
Introduction to Forest Ecosystem Services and their Institutional Framework

Forests are delivering multiple benefits for society in the form of goods and services such as wood, food, clean water, energy, protection from floods and soil erosion, climate regulation or regulation of hydrological cycles, recreation and cultural values. These benefits are known as forest ecosystem services (FES). They play an important role in human well-being, make significant direct and indirect contributions to national economies and contribute to environmental stability (Alcamo et al. 2005).

The Millennium Ecosystem Assessment (Alcamo et al. 2005) was one of the first important global studies on ecosystem services (ES) and its framework is widely accepted and seen as a useful starting point (Wallace 2007; Boyd, Banzhaf 2007; Fisher et al. 2009). More recently the Mapping and Assessment on Ecosystems and their Services – MAES framework (Maes et al. 2013) was adopted by Member States of the European Union. It builds on the findings of the Millennium Ecosystem Assessment and global initiative the Economics of Ecosystems and Biodiversity (TEEB 2010) and was further refined as an operational framework at European level.

The MAES framework (Maes et al. 2013) is successful in integrating the biophysical domain with the socio-economic drivers affecting ES and with the role of biodiversity in ecosystem functions and services (Figure 1). Therefore, we consider this framework as a robust basis for studying European forests in terms of ecosystem service delivery and opportunity.

Maes et al. (2013) define ecosystem services as the benefits that people obtain from ecosystems - the direct and indirect contributions of ecosystems to human wellbeing. The concept "ecosystem goods and services" is synonymous with ecosystem services.

MAES, according to the Common International Classification of Ecosystem Services (CICES v4.3), classifies ecosystem services (ES) into three groups: Provisioning, Regulation and Maintenance, and Cultural services. However, there are also other two international classification systems of ES, i.e. classifications according to the Millennium Ecosystem Assessment (2005) and TEEB initiative (2010). A comparison of these three main classification schemes was addressed by the former FOREST EUROPE Expert Group on Valuation of Forest Ecosystem Services during 2013–2014. The results of the Expert Group can be found in the FOREST EUROPE Final Report on Valuation of FES (2014). Additionally, Maes et al. (2013) provides a comparison of the three classification schemes and discusses the adoption of CICES.

Ecosystems are shaped by the interaction of communities of living organisms with the abiotic environment. Biodiversity, the variety of all life on earth, plays a key role in the structural and functional set-up of ecosystems, and is an essential feature to maintaining basic ecosystem processes and supporting ecosystem functions. Ecosystem functions are defined as the capacity or the potential to deliver ecosystem services. Ecosystem services are, in turn, derived from ecosystem functions and represent the realized flow of services for
which there is demand. For the purpose of this framework, ecosystem services also encompass the goods derived from ecosystems. People benefit from ecosystem services. These benefits are, among others, nutrition, access to clean air and water, health, safety, and enjoyment and they affect human wellbeing which is the key target of managing the socio-economic systems. The focus on benefits implies that ecosystem services are open to economic valuation. However, not all benefits to people from ecosystems can be measured in monetary terms. Therefore, it is important to include other values as well, such as health value, social value or conservation value. The governance of the coupled socio-economic-ecological system is an integral part of the framework: Institutions, stakeholders and users of ecosystem services affect ecosystems through direct or indirect drivers of change. Policies concerning natural resource management aim to affect drivers of change to achieve a desired future state of ecosystems. Many other policies also affect these drivers and thus can be added to the framework as they have an impact on forest ecosystems even though they might not target them at all (e.g. through the construction of roads or other infrastructure, or industrial policy through pollution) (Maes et al. 2013).
References


PART I.

Valuation of Forest Ecosystem Services
The value of ecosystems and their services is viewed and expressed differently by different disciplines, cultural conceptions, philosophical views, and schools of thought (Goulder, Kennedy 1997). The basis for the provision of ES is represented by natural capital, which is defined by the Natural Capital Committee as “those elements of the natural environment which provide valuable goods and services to people” (NCC 2017). Assessing the value of natural capital (elements of the natural environment which provide valuable goods and services to people), is fundamental to deciding how and where funds should be allocated to restore, maintain and manage the natural environment (NCC 2017).

The ecosystem services approach characterises the environment as a complex natural factory (Figure 2) engaged in a myriad of productive processes. Unlike the productive activities of firms and households, the productive processes in the environment are not organised by humans but arise spontaneously in nature. In an exact parallel to the human economy, the productive activities of nature are described by environmental production functions. Just like their human controlled counterparts, environmental production functions require inputs and deliver outputs - ecosystem services (Binner et al. 2017).

Economic valuation is used as a tool that enhances the ability of decision-makers to evaluate trade-offs between alternative ecosystem management regimes and courses of actions that alter the use of ecosystems and the services they provide (Alcamo et al. 2003).

The economic valuation of ES can provide information for decision makers at many different levels (Turner et al. 2010). This ranges from national and international policy decisions to regional and sub-regional decisions and local planning decisions and projects. The challenge in each case is to identify all the ES that will be affected by the decision and to obtain sufficient information to conduct the ecosystem service assessment, including linking the assessment of changes in service provision to measures of changes in human welfare (DEFRA 2007).

There are a number of other potential reasons for undertaking valuation of FES. The most common are as follows (Merlo et al. 2005; FOREST EUROPE 2014; Mavsar, Varela 2014):

- to assess (and improve) the overall contribution of forests ecosystems to social and economic well-being,
- to obtain information about the relative importance of FES and preferences for their provision across and from different stakeholder groups and understand how and why stakeholders use forests as they do,
- to assess the relative impact of alternative actions, as a decision support tool,
- to identify potential winners and losers when adopting a certain management alternative,
- evaluating the impacts of environmental policies,
- establishing incentive schemes or markets for FES.

1. **Introduction to Valuation of Forest Ecosystem Services**
Figure 2 – The natural factory (Binner et al. 2017)

It should be also noted that economic valuation can be an important step for developing mechanisms to measure the benefits of the services and in establishing finance/incentive systems such as payments for ecosystem services (PES). PES covers a variety of financing arrangements through which the beneficiaries of ES pay the provider of those services (Gutman 2006), thus offering incentives for protecting or restoring of ecosystems and thus maintaining or increasing the supply of such services.

A major challenge regarding the valuation of FES is that many of the services provided are not traded in markets, making it difficult to observe their values directly (FOREST EUROPE 2014). Many FES are provided as public goods although, in practice, most are not pure public goodsassin but display some of associated characteristics (they may be enjoyed by any number of people without affecting other people’s enjoyment). An issue with public goods is that, although people value them, there are no incentives to pay to maintain their provision (Nasi et al. 2002; Šišák 1996, 1997, 1998, 2011).

When public goods and services are supplied to either society or specific groups of users for free, or at a price which is below the production costs of equivalent goods and services, forest owners receive little or no monetary incentive to conserve forest ecosystems in good conditions and maintain or improve the provision of the services. This can result in declines in both the quantity and quality of the services. Possible solutions include applying regulations to enforce their provision or developing incentive mechanisms (including market-based instruments) which encourage forest owners to provide them. Therefore, knowledge of how to estimate the economic value of the services is often a crucial step in
providing evidence to support the introduction of such mechanisms (FOREST EUROPE 2014). Economic valuation in this sense relates to the demand side, i.e. preferences of the group of ecosystem service users, or society as a whole. For the supply side (forest enterprises or owners), cost based methods are used. As soon as both kinds of information are available, it is possible to determine a value range within which one can establish “efficient” incentives for forest owners.

According to Binner et al. (2017) the concept of economic value is based on the idea that value (or utility) is a human construct and that it provides a measure by which we might gauge what is the best for a human society. It is compatible with the idea that value may come from non-human entities, but only insomuch as they increase the well-being experienced by humans either by supporting our livelihoods, enhancing our existence or because of a sense of moral duty. Binner et al. (2017) state that the value flow from a FES is determined by at least two things:

- the FES’s attributes, as determined by the environmental production function through which it is delivered,
- the context within which the FES are consumed, as determined by the other FES, FES and qualifiers that enter the human production function through which the FES delivers value.

Also the issue of aggregation has to be addressed in determining the economic value of a FES, because we need to add together the value flows accruing to all the individuals who gain benefit from that FES (Binner et al. 2017).

A concept which is important in this context is that of the Total Economic Value (TEV). It has been developed in order to consider values, including non-use values, systematically and comprehensively.

The TEV approach is based on the different benefits that humans may obtain from forest ecosystems (Figure 3). The main aim of TEV classification, used in Pearce and Moran (1994) and Merlo et al. (2005), was to assess the overall contribution of forest ecosystems to “social and economic well-being.”

This framework typically disaggregates TEV into two main categories: use values and non-use values (Pearce 1991; Groombridge 1992). Traditionally the distinction between use and non-use values has been characterised as the difference between a value that is derived from physical interaction with a FES (use value) and one in which value is derived without physical proximity to or interaction with a FES (non-use values) (Binner et al. 2017).

First, use value refers to the value of ES that are used by humans for consumption or production purposes. It includes tangible and intangible services of ecosystems that are either currently used directly or indirectly, or that have a potential to provide future use values. The TEV separates use values as follows (DeFries, Pagiola 2005; Binner et al. 2017):

- direct use values are derived from FES that are used directly by humans. They include the value of consumptive and non-consumptive uses and they are typically enjoyed by people located in or

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**Figure 3** - The Total Economic Value (TEV) of the benefits of forest functions (Pearce, Moran 1994; Merlo et al. 2005)
visiting the ecosystem itself. In other words, an environmental good or service generates direct value if it enters the human utility function as a FES.

- indirect use values are derived from a wide range of FES that provide benefits outside the ecosystem itself. That means an environmental good or service generates indirect value if it contributes, through some biophysical process in an environmental production function, to the supply of some other FES.

- the notion of option values, introduced by Weisbrod (1964), are derived from preserving the option to use the services, which may not be used at present, in the future (Krutilla, Fisher 1975). Quasi-option value is a kind of value that refers to the value of information secured by delaying a decision, where outcomes are uncertain and where there is opportunity to learn by delay. This is to say that the information on value will only be revealed over time, mainly because there is uncertainty about the future value of a natural resource (Arrow, Fisher 1974). As Binner et al. (2017) state, the distinction between direct and indirect values is important because it informs us as to when we can value an environmental good or service directly (as a FES) as compared to when we first have to understand the biophysical process supporting the provision of FES.

Second, non-use values from ecosystems are those values that do not involve direct or indirect uses of the ecosystem service in question. Humans ascribe value to knowing that a resource exists, even if they never use that resource directly. Individuals reflect satisfaction derived from the knowledge that ES are maintained and that other people have or will have access to them (Kolstad 2000). Two typologies of non-use values are usually referred to as bequest values associated with altruist values, and existence values. Non-use values are the hardest and the most controversial to estimate. These values involve greater challenges for valuation because they can be related to moral, religious or aesthetic properties, for which markets usually do not exist. This is different from other services’ values,
which are associated with the production and valuation of tangible things or conditions. Non-use values in general involve the production of experiences that occur in the valuer’s mind. These services are therefore co-produced by ecosystems and people in a deeper sense than other services (Chan et al. 2006).

The study on the TEV of Mediterranean Forest (Merlo et al., 2005) is considered the first attempt for a comprehensive and systematic evaluation of FES in Europe (Mediterranean countries). This study filled a knowledge gap regarding the valuation of non-wood forest products (NWFPs) and provided a first estimate to the TEV including both NWFPs and wood forest products into a common framework.

In conclusion, as Binner et al. (2017) pointed out, the various categorisations of ES values (such as that under TEV) are just proposals and that to a certain extent those attempts are superseded by the ecosystem services approach that focus on environmental goods and services as arguments in human production functions. In short, an environmental good or service may generate as many different values as human production functions to which it contributes.

1) A scientific term for those goods is „club goods“ (e.g. watershed protection services, ecotourism in protected areas, hunting clubs) (Thorsen et al. 2014)

2) Aggregation means how many people enjoy value and how this is mediated by proximity. The context within which individuals consume a particular FES will differ, perhaps markedly, across individuals. The proximity of a woodland used for recreation will differ across individuals, changing the costs they must incur in accessing that woodland and the proximity of other natural areas offering substitute locations for outdoor recreation.
In the last decades, valuation methods (VM) have reached a considerable degree of sophistication. This period has also witnessed a gradually emerging consensus on the state-of-the-art of the range of valuation methods at hand, which is reflected by the fact that recent handbooks and manuals on the topic provide very similar overviews and assessments of the individual tools, with differences remaining essentially on the level of terminology and classifications (Pagiola et al. 2004; de Groot et al. 2006; FOREST EUROPE 2014; Plan Bleu 2015; DEFRA 2007).

The scientific literature on valuation and assessment of ES is based on two distinct foundations. The ecological evaluation methods aim to assess the significance of biophysical phenomena for ecosystems, without necessarily focusing at consumer preferences. The ecological evaluation methods do not place monetary values on ES. The economic valuation methods focus on market and non-market values of ES for people, rather than on the complex internal structure of ecosystems. The methods for economic valuation of ES are not restricted to monetary valuation alone; they include the analysis of any trade-off made by people between the different possibilities of resource use. A detailed historical overview can be found in e.g. Gómez-Baggethun et al. (2011) or Liu et al. (2010).

Figure 4 captures detailed classification of three main categories of economic valuation methods: (i) revenues oriented, (ii) cost based, and (iii) non-monetizing. In this regard, a spectrum of non-market valuation techniques has been developed to value ES. The use of a monetary metric assumes that individuals are willing to trade the ecosystem service being valued for other services represented by the metric. The basic distinction among monetary valuation methods is based on the data source, that is, whether it derives from observations of human behaviour in the real world (“revealed preferences”) or from human responses to hypothetical questions (“stated preferences”) (Liu et al. 2010).

Economic valuations usually attempt to elicit individual preferences within the general public for changes in the provision of services or in the state of the environment in monetary terms. These are based on the fundamental principles of welfare economics; whereby the changes in the well-being of individuals are reflected in their willingness to pay or willingness to accept compensation for changes in their level of use of a particular service or bundle of services (Hanley et al. 2001). In accordance to DEFRA (2007), the main types of economic valuation methods available for estimating public preferences for changes in ES are as follows (see Figure 5):

- **Revealed preference methods** are based on actual observed behaviour data, including some techniques that deduce values indirectly from behaviour in surrogate markets, which are assumed to have a direct relationship with the ecosystem service of interest.

- **Stated preference methods** use carefully structured questionnaires to elicit individuals’ preferences for a given...
Figure 4 - Overview of valuation approaches and methods
Figure 5 - Economic valuation methods available for estimating individual preferences
change in a natural resource or environmental attribute. Stated preference methods are based on hypothetical rather than actual data on behaviour; for the former the value is inferred from people’s responses to questions describing hypothetical markets or situations.

2.1 Ecological Evaluation Methods – Mapping and Assessment

Ecological evaluation methods are used to quantify the amount of quality of ES. Furthermore, the outcomes of this evaluation is frequently used as input for economy valuation (e.g. to understand the status quo and changes in ES provision).

Ecological evaluation methods use landscape indicators reflecting the provision of ecosystem services, or the condition of ecosystems, using a series of metrics. These methods are based on an analysis and synthesis approach to landscape evaluation, and provide a suite indicators representing landscape traits. Ecological evaluation is difficult because of the large number of traits that can be assessed, issues of data availability and scale of assessment. For example, landscape ecology indexes provides an valuable option, they focus on a quantitative evaluation using a discrete territory classification usually represented in maps using indicators (Gomez-Sal et al. 2003; Chan et al. 2006; de Groot et al. 2010; Spangenberg, Settele 2010; Baveye et al. 2013).

Several evaluation methods are used as described below.

**Diversity indices.** A diversity index is a mathematical measure of the species diversity in a community. Diversity indices provide more information about community composition than simply species richness (i.e. the number of species present); they also take the relative abundance of different species into account. The most commonly used are (Magurran 1988; Roth et al. 1994; Begon et al. 1996; Chao, Shen 2003; Sarvašová et al. 2014):

- **The Shannon–Wiener Diversity Index** enables to measure the diversity of the landscape. Its calculation takes into account the proportional representation of each type of land cover. The rate of the index increases when the number of various land cover types increases and/or if the proportional distribution of various land cover types is balanced.

- **Simpson’s Index** is often used to quantify the biodiversity of a habitat. It takes into account the number of species present, as well as the quantity of each species. It measures the probability that two individuals randomly selected from a sample will belong to the same species.

- **The Berger–Parker Diversity Index** is the proportion of the most common species in the community or sample. It is the simplest and most easily understood diversity index, since it only calculates the proportion of the commonest species in a sample.

- **Similarity indices** measure the similarity between communities based on species composition and are useful in comparing communities under different forms of management.

The evaluation of the functional efficiency of non-forest vegetation. Non-forest vegetation is one of the significant components of the landscape structure, and is important for humans, plant and animal species. Non-forest vegetation includes original, natural, semi-natural and synantrop or man-made communities. In the agricultural and urbanized landscape, associations support the provision of ES and positively influence ecological stability. This method based on site conditions and relations to ecological stability has been used by various authors e.g. Múdry (1983); Ulrychová (1995); Rózová (1994); Sláviková (1987,1990).

**Landscape complexes** propose FES valuation of the landscape on two levels (Michal 1997a,b):

1. typological classification (based on geo-ecological criteria) focuses on the landscape as a set of ecosystems and how the history of ecosystems influenced by mankind is reflected in the landscape shape and arrangement;
2. land use, which is ecological and aesthetic classification of ecosystems in a specific sense (foremost according to the environmental values or from the landscape scenery). Aesthetic valuation criteria are in principal social and take into account the share of subjective processes in phenomena evaluation.
Typology of the area leads to the calculation of the ecological stability coefficient for individual cadastral areas for the purposes of lands consolidation according to the degree of their ecological stability. In general, land use complexes with a low landscape values have a high need for new positive ecological and aesthetic values and thus greater ES.

**Landscape Ecological Planning (LANDEP).** This method calls fora multidisciplinary approach to landscape evaluation as an area in which the human activities are developed on the basis of natural phenomena and processes. LANDEP is a synthesis of knowledge of potential possibilities of ecologically optimum landscape utilization from the viewpoint of purposeful formation of the conditions for the conservation and development of healthy populations of organisms and humans and for the development of human society. From the methodological viewpoint, LANDEP is based on the analysis, interpretation, synthesis and evaluation of ecological features. The result of LANDEP is a proposal for ecologically optimum landscape utilization aimed at the harmonization of social activities in the landscape with its ecological features in time and space (Ružička, Miklós 1982; Karvonen 2000; Pitkänen et al. 2000; Sanderson et al. 2002; Mörtberg et al. 2007).

A series of approaches for mapping and assessment of ecosystem services has been proposed in recent years. Summaries of these approaches are presented in Eigenbrod et al. (2010); Ayanu et al. (2012); Crossman et al. (2012); Egoth et al. (2012); Maes et al. (2012); Martínez-Harms and Balvanera (2012); Crossman et al. (2013); Schägner et al. (2013); and Willemen et al. (2015). Maps of ecosystem services and ecosystem condition are a popular tool for policy support and territorial assessment. They provide measures of ecosystem services provision, trade-off between services and can suggest provision hotspots and coldspots.

The typology of methods is built upon the studies of Martínez-Harms and Balvanera (2012), Eigenbrod et al. (2010) and Schägner et al. (2013), and the summary of Barredo et al. (2015) where six categories are described:

- **Look-up tables (LUT).** This approach makes use of existing ecosystem services values from the literature and applies to land-cover classes in other spatial domains. This is the Tier 1 approach according to MAES (Maes et al. 2014).
- **Expert knowledge.** Experts rank land-cover types based on their potential to provide specific ecosystem services. This method is founded in expert knowledge with regards to the potential of the land cover categories to supply given ecosystem services. Often this approach is integrated with the LUT approach.
- **Causal relationships.** This approach incorporates existing knowledge about how different layers of information relate to ecosystem processes and services to create a new proxy layer of the ecosystem services. This includes methods using spatial variables such as distance relationships (e.g. to roads), amount of a type of land (e.g. protected areas or forest areas), land cover data, population density, climate data, soil data, elevation. Causal relationships are usually taken from the literature, expert knowledge or are derived empirically on the basis of available observational data; then spatially-explicit variables are integrated using GIS modelling tools for producing a new proxy layer (map) of ecosystem services. This is the Tier 2 approach according to MAES (Maes et al. 2014).
- **Statistical and machine learning models.** This approach employs field data of ecosystem services for modelling the relationship with explanatory variables and proxies, e.g. biophysical data and other sources of information obtained from GIS. One of the strengths of this approach is the ability to provide measures of error/accuracy. This is the Tier 3 approach according to MAES (Maes et al. 2014).
- **Implicit modelling.** This approach uses value functions relating variation in ecosystem services values to variation in the characteristics of the ecosystem, context and beneficiaries of the services. Local-level parameter values are input into the value function to extrapolate the value to other sites of the study area with unknown value information. This approach is common in studies from the environmental economics domain (e.g. for mapping cultural services, but not only) (Schägner et al. 2013).
Representative sampling. This approach offers the best estimate of observed levels of ecosystem services. However, ecosystem services mapping studies based on this approach are limited due to the high costs and difficulty to collect the large amount of data required (Eigenbrod et al. 2010).

2.2 Economic Valuation Methods

Economic values have been associated with forestry in the past, in keeping with the traditional orientation of forest management, which was towards production of timber and other products for the market. Market prices were thus considered to be the source of information determining the value of forest wood products. Everyday language often confuses “economic values” with “market prices”. However, both terms are not synonymous. First, market prices only tell us the minimum amount that people who buy a good or service are willing to pay for it, whereas actual prices do not reflect the fact that people might spend more for this good or service than required by the market. Second, prices might be distorted by market imperfection due to external effects or other anomalies.

There has been an increased demand for a wide variety of goods (foods, fuels, medicines, fodder, etc.) and services (erosion protection, fresh water, options for recreation and aesthetics, habitat provision for biodiversity, climate regulation, air quality regulation, etc.) by humans; however, market based methods cannot capture many of those goods and services, which are non-marketed, or relate to benefits derived outside the forestry sector or to influences on it that are external. The supply of non-market goods or services should be assessed at the local level, but in contrast the drivers affecting forest ecosystems (and hence the services) may be of wider nature, such as national, transboundary regional or global level.

There are no absolute values for non-market forest services, as they are based on perceptions by individuals and groups, which are subject to dynamic changes in their situation, needs and aspirations. These perceptions in the particular case of forests have been evolving rapidly in recent years with the broadening of interests, the increasing number of interest groups, the diversity of perceptions by different groups and the awareness of the wide range of goods and services provided by forests at local, regional, national and global levels. Furthermore, provision of goods and services involve costs and benefits whose distribution among interest groups is often a significant element in terms of its political nature and decision making processes. The social and environmental impacts may also change rapidly and the direction of change may be different for the various groups affected (UNECE 1996).

Valuation might also be useful for assessing potential levels of compensation to forest managed with conservation objectives or to loss of income due to limits in the use of the full production potential. The valuation approach will therefore be shaped by the decision-making context within which the information is to be used and will focus on answering the basic questions when comparing a proposed change with the status quo (UNECE 1996). Knowing the value of the ecosystem services provided by a forest could also simply confirm or even reinforce the need to restore, protect and conserve it and change people’s perception of that forest.

In the following section, different approaches and methods for economic valuation of forest ecosystem services are broadly described. Examples on how these methods were applied for valuation of forest ecosystem services can be found at the FOREST EUROPE Web-based Portal on FES https://foresteurope.org/valuation-forest-ecosystem-services/.

2.2.1 Revenues Oriented Valuation Methods

Contingent Valuation Method (CVM) is a questionnaire based technique that seeks to discover individual preferences for an environmental change. It uses one of two measures of consumer’s surplus: compensating variation (CV) or equivalent variation (EV). CV is the amount of money (change in income) necessary to make an individual indifferent with respect to an initial situation and a new situation with different prices. EV may be viewed as a change in income equivalent to a change in welfare after a change in prices has occurred. CVM is used to estimate the consumer’s willingness to pay (WTP) for a specified good or service, or his/her willingness to accept compensation (WTA) for forgoing a desired
good or service. In practice, it is usually derived from the responses of potential consumers to a hypothetical exchange situation (UNECE 1996).

The method assumes that the consumer's expressed WTP in a hypothetical situation is a utility indicator to the consumer in an actual situation. The basic premise of the contingent valuation method is that individuals are sensitive to a given environmental change and that their preferences could be measured in terms of their WTP to undergo this change (or their WTA to avoid it). Therefore, the given change is presented to individuals through a survey where the environmental change is presented and where people are directly asked to state their WTP or their WTA for the given environmental change (Plan Bleu 2015).

- **Suitability for the FES to be valued** – all forest services
- **Benefits of the method** (FOREST EUROPE 2014; Plan Bleu 2015):
  - Measurement of non-use values is possible (to provide a comprehensive measure of TEV);
  - Valuation of future goods and services possible;
  - The use of surveys allows to collect relevant socioeconomic and attitudinal data on the respondents that could be relevant for understanding the variables influencing social preferences and choices;
  - The use of surveys allows to estimate hypothetical changes and their impact before they have taken place;
  - Participative/deliberative approaches before valuing the service at stake seem to provide with more stable results.
- **Limitations of the method** (FOREST EUROPE 2014; Plan Bleu 2015):
  - Results are sensitive to numerous sources of bias in survey design and implementation;
  - Preferences for non-use values tend to be less stable;
  - Budget and time demands are high.

The most used variants of CVM are (Bateman, Turner 1992, 1993):

- **Open-ended** – respondents are asked how much they are willing to pay. So, the answer is an amount of money at a continuous scale. The resulting distribution of this continuous variable can be described by its moments (e.g., mean WTP and variance), and may further be analysed using ordinary regression techniques (e.g., ordinary least squares).
- **Dichotomous choice** – instead of open questions respondents are asked whether they would pay a certain amount. Different respondents are asked for different amounts, which are randomly distributed over the sample. Dichotomous choice allows only for “yes” and “no” answers. Questions can be single-bounded, where only one question is asked, or multiple-bounded, where follow-up questions with higher or lower amounts, depending on the initial reply, are asked. WTP is estimated using logit or probit techniques.
- **Iterative bidding game** – submit to respondents different rounds of discrete choice questions or bids, with a final open-ended WTP question. They may suffer from lack of incentive compatibility and starting point bias, and fatigue effects.
- **Payment card** – card indicates range of possible values, one of which is pointed out by respondent. Instead of starting point bias, payment cards may suffer from range bias, as the respondents might misunderstand the amounts presented in the payment card as hints to an “acceptable” answer.

**Choice modelling** attempts to determine the WTP of an individual by analysing his choices between different alternatives (Hanley, Wright 1998; Philip, MacMillan 2005). Individuals are faced with two or more alternatives with shared attributes of the services to be valued, but with different attribute levels (one of the attributes being the money people would have to pay for the service). The alternatives are designed so that the respondent’s choice reveals the marginal rate of substitution between the attributes and the item that is traded off (e.g. money) (Pascual, Muradian 2010).

The basic premise of the choice experiment is that a forest good or service can be decomposed in a bundle of attributes or features, and that
individuals are sensitive to changes in these attributes.

- **Suitability for the FES to be valued** – all forest services
- **Benefits of the method** (FE 2014; Plan Bleu 2015):
  - Measurement of non-use values is possible (to provide a true measure of TEV);
  - Valuation of future services is possible;
  - Valuation of several services at the same time (including their trade-offs);
  - The use of surveys allows to collect relevant socioeconomic and attitudinal data on the respondents that could be relevant for understanding the variables influencing social preferences and choices;
  - The use of surveys allows to estimate hypothetical changes and their impact before they have taken place;
  - Participative/deliberative approaches before valuing the service at stake seem to provide with more stable results.

- **Limitations of the method** (FOREST EUROPE 2014; Plan Bleu 2015):
  - High data requirements;
  - Analysis mathematically complicated;
  - Interpretation is not straightforward for lay people;
  - Preferences for non-use values tend to be less stable;
  - Budget and time demands are high;
  - High risk of biases that may lead to inaccurate WTP estimations.

The most used variants of choice modelling are as follows (Hanley et al. 2001):

- **Contingent ranking** – respondents are required to rank a set of alternative options, characterised by a number of attributes, which are offered at different levels across options. A status quo option is normally included in the choice set to ensure welfare consistent results.

- **Contingent rating** – exercise respondents are presented with a number of scenarios and are asked to rate them individually on a semantic or
numeric scale. This approach does not involve a direct comparison of alternative choices and consequently there is no formal theoretical link between the expressed ratings and economic choices.

- **Pair comparisons** - exercise respondents are asked to choose their preferred alternative out of a set of two choices and to indicate the strength of their preference in a numeric or semantic scale. This format is also known as graded or rated pairs.

- **Polychotomous choice** - respondents are presented with a series of alternatives, differing in terms of attributes and levels, and asked to choose their most preferred. A baseline alternative, corresponding to the status quo or “do nothing” situation, is usually included in each choice set. This is because the options must always be in the respondent’s currently feasible choice set in order to be able to interpret the results in standard welfare economic terms.

**Averting behaviour models** are based on the presumption that people will change their behaviour and invest money to avoid an undesirable outcome. Thus, averting behaviour analyses the rate of substitution between changes in behaviour and expenditures on and changes in environmental quality in order to infer the value of certain non-marketed environmental attributes (Dickie 2003). These models are similar to the travel cost method and hedonic pricing (described on p. 30–31) but they differ as they use as a basis individual behaviour to avoid negative intangible impacts as a conceptual base. However, the situation is complicated by the fact that these market goods might have more benefits than simply that of reducing an intangible bad. Averting behaviour occurs when individuals take costly actions to avoid exposure to a non-market bad. It is needed to take account the fact that valuing these alternative actions might not be a straightforward task, for instance, if time which would have been spent doing one thing is instead used to do something else, not only avoiding exposure to the non-market impact in question, but also producing valuable economic outputs (Hadley et al. 2011).

- **Suitability for the FES to be valued** - regulating services
- **Benefits of the method:**
  - It has a sound theoretical basis;
  - It uses data on actual expenditures and data requirements can be modest.
- **Limitation of the method:**
  - Not a widely used methodology;
  - It can only estimate use values;
  - Limited to cases where households spend money to offset environmental hazards/nuisances;
  - Confined to cases where those affected are aware of the environmental issue and act because of them;
  - Appropriate data may be difficult to obtain.

**Value Transfer Method** (also referred to as Benefit transfer) is not a direct valuation method, because it is based on the transfer of valuation. It is used to estimate economic values for ES by transferring the available information from studies already completed in another location and/or context (Navrud, Ready 2007; Sarvašová et al. 2014). Thus, the basic goal of benefit transfer is to estimate benefits for one context by adapting an estimate of benefits from some other context (Plan Bleu 2015).

- **Suitability for the FES to be valued** - all forest services, however, it was shown that it is more reliable for transferring use values.
- **Benefits of the method** (Plan Bleu 2015):
  - Typically less costly than conducting an original valuation study;
  - Economic benefits can be estimated more quickly when undertaking an original valuation study;
  - The method can be used as a screening technique to determine if a more detailed, original valuation study should be conducted.
- **Limitations of the method** (Plan Bleu 2015):
  - Value transfer may not be accurate, except for making gross estimates of values, unless the sites share all of the site, location, and user specific characteristics;
  - Good studies for the policy or issue in question may not be available;
It may be difficult to track down appropriate studies, since many are not published;
Reporting of existing studies may be inadequate to make the needed adjustments;
Adequacy of existing studies may be difficult to assess;
Extrapolation beyond the range of characteristics of the initial study is not recommended;
Value transfers can only be as accurate as the initial value estimate;
Unit value estimates can quickly become dated.

There are two main forms of the value transfer method (Navrud, Ready 2007):

- **Unit Value Transfer** - it is the simplest method which builds on the transfer of actual value estimates from other studies, appropriately adjusted for inflation, the differences in purchasing power of income across regions and, in some cases, the income variation.

- **Function Transfer Approach** - it is more ambitious and suggests transferring value functions from other studies. The benefit function statistically relates people willingness to pay to ecosystem characteristics and the people whose values were elicited.

**Market price method** estimates the economic value of ecosystem services that are bought and sold in markets. The market price method can be used to value changes in either the quantity or quality of a service. It uses standard economic techniques for measuring the economic benefits from marketed services, based on the quantity people purchase at different prices, and the quantity supplied at different prices. The standard method for measuring the use value of resources traded in the marketplace is the estimation of consumer surplus and producer surplus using the market price and quantity data (UNECE 1996; Sarvašová et al. 2014; Plan Bleu 2015).

- **Suitability for the FES to be valued** - all marketed services
- **Benefits of the method** (FOREST EUROPE 2014; Plan Bleu 2015):
  - People’s values are likely to be well-defined as it reflects an individual willingness to pay for services that are exchanged in markets. Unless there are known distortions to correct for, such market prices can be taken as economic values (due to reflecting the minimum willingness to pay of consumers);
  - Data availability;
  - Uses observed data of actual consumer preferences;
  - Uses standard, accepted economic techniques.

**Limitations of the method** (FOREST EUROPE 2014; Plan Bleu 2015):

- Market data are available only for a limited number of services;
- True economic value of services may not be fully reflected in market transactions;
- Seasonal variations and other effects on price must be considered;
- Cannot be easily used to measure the value of larger scale changes that are likely to affect the supply of or demand for a service;
- Usually, the market price method does not deduct the market value of other resources used to bring ecosystem products to market because consumer surplus and producer surplus are not estimated, and thus may overstate benefits.

**Efficiency (shadow) prices.** The market price does not necessarily mean the “proper” price and/or reflect the true economic efficiency price. There are market and policy failures that can distort market prices. Market failures refer to the inability of market prices, under certain conditions, to reflect accurately the value of environmental services. Policy failures concern instances where government policies have unintended effects, or sometimes even side-effects or cause resource-use behaviour inappropriate from a societal perspective.

In financial analysis, no account is taken of any of these failures that distort market prices. Therefore, it is advisable to look at their economic value in order for their value to society as a whole to be reflected, as in the case, for example, of
alternative forest land uses. To do so, the market price is adjusted. There are various methods for correcting market and policy distortions. A variant of the market price-based method uses shadow prices (market prices adjusted for transfer payments, market imperfections and policy distortions). Shadow prices may also incorporate distribution weights, where equality concerns are made explicit (UNECE 1996).

**Suitability for the FES to be valued:** services that are traded in domestic or international markets and it may also be calculated for non-marketed services

**Benefits of the method:**
- Reflect the true economic value (opportunity cost) to society as a whole;
- Especially useful when there are significant distortions on market prices (considering that prices often reflect the effects of subsidies given to foresters/farmers or of trade policies).

**Limitations of method** (UNECE 1996):
- Derivation of shadow price is complex and may require substantial data;
- Decision-makers may not accept what they might consider to be artificial prices;
- Market prices are often more readily accepted by decision makers than artificial values derived by the analyst;
- Market prices are generally easy to observe, both at a single point and over time;
- Market prices reflect the decision of many buyers, whereas calculating shadow prices often relies on assumptions and assessments of the analyst;
- The procedures for calculating shadow prices are rather imperfect and therefore estimates can, in certain cases, introduce larger discrepancies than even the simple use of imperfect market prices;
- Each case should be analysed within the context in which the valuation is being made, and should take into account the data and resource constraints. There cannot be a simple blueprint for every case.

**Travel Cost Method (TCM)** derives willingness to pay for environmental benefits at a specific location by using information on the amount of money and time that people spend to visit the location. It is based on the rationale that recreational experiences are associated with cost (direct expenses and opportunity costs of time). The value of a change in the quality or quantity of a recreational site (resulting from changes in biodiversity) can be inferred from estimating the demand function for visiting the site that is being studied (Bateman et al. 2002; Kontoleon, Pascual 2007).

This method estimates peoples’ willingness to pay from observations of the number of trips that they make at different travel costs. This is analogous to estimating peoples’ willingness to pay for a marketed good based on the quantity demanded at different prices (Plan Bleu 2015).

**Suitability for the FES to be valued** – recreational services

**Benefits of the method** (FOREST EUROPE 2014; Plan Bleu 2015):
- Similar to more conventional approaches to estimate economic values based on market prices;
- Based on actual behaviour rather than on hypothetical behaviour of the respondents;
- On-site surveys provide opportunities for large sample sizes;
- Results are relatively easy to interpret and explain;
- Relatively inexpensive to apply.

**Limitations of the method** (FOREST EUROPE 2014; Plan Bleu 2015):
- Opportunity costs of time (i.e. the idea that time spent traveling could have been used in other ways) are difficult to determine and might not be empirically observable at all, which requires additional assumptions;
- Assumption that people respond to changes in travel costs the same way that they would respond to changes in admission price might not always be true;
- Limited in its scope of application because it requires user participation;
- Standard approaches provide information about current conditions,
but not about gains or losses from anticipated changes in resource conditions;

- The simplest travel cost models assume that individuals take a trip for a single purpose;
- The availability of substitute sites will affect values.

Modern variants of travel-cost analyses are based at Random utility theory. **Random utility models** arise from the empirical assumption that people know their preferences (utility) with certainty, but there are elements of these preferences that are not accessible to the empirical observer (Herriges, Kling 1999; Parsons 2003). Thus, parameters of peoples’ preferences can be recovered statistically up to a random error component. This econometric approach is used to estimate modern travel-cost models.

- **Suitability for the FES to be valued** - recreational services
- **Benefits of the method** (NRC 2004):
  - Uniquely designed to estimate values for attributes of recreation sites, which include the quantity and quality of the ES;
  - The best approach to use to estimate benefits for specific characteristics, or quality changes, of sites, rather than for the site as a whole;
  - The most appropriate approach when there are many substitute sites.
- **Limitations of the method** (Grüningen 2016):
  - Data needed not only from one site but also concerning all other sites;
  - Econometric models are more complex.

**Hedonic pricing method** - relies on market transactions for differentiated services to estimate the economic benefits or costs associated with environmental quality. The basic premise of the hedonic pricing method is that the price of a marketed service is related to its characteristics, or the services it provides (Rosen 1974). It is based on the assumption that services can be considered aggregates of different attributes, some of which, as they cannot be sold separately, do not have an individual price (FOREST EUROPE 2014).

It utilizes information about the implicit demand for an environmental attribute of marketed commodities. For instance, houses or property in general consist of several attributes, some of which are environmental in nature, such as the proximity of a house to a forest or whether it has a view on a nice landscape. Hence, the value of a change in biodiversity or ES will be reflected in the change in the value of property (either built-up or land that is in a (semi-) natural state). By estimating a demand function for property, the analyst can infer the value of a change in the non-marketed environmental benefits generated by the environmental good (Plan Bleu 2015).

- **Suitability for the FES to be valued** - air pollution, water pollution, cultural services (aesthetic views) and recreational services
- **Benefits of the method** (FOREST EUROPE 2014; Plan Bleu 2015):
  - May be conducted with already existing data;
  - Can be used to estimate values based on actual choices;
  - Property markets are relatively efficient in responding to information, so can be good indications of value;
  - The method is versatile, and can be adapted to consider several possible interactions between market services and environmental quality;
  - Property records are typically very reliable.

**Limitations of the method** (FOREST EUROPE 2014; Plan Bleu 2015):

- It can be applied only in presence of a good number of market exchanges, as the model representing the market requires a certain number of good quality data;
- The market must be sufficiently transparent;
- Scope of environmental benefits that can be measured is mainly limited to things that are related to housing prices, it is not possible to estimate the TEV of the environmental service, but only the value connected to present and, with some caution;
Only captures people’s willingness to pay for perceived differences in environmental attributes, and their direct consequences;

Assumes that people have the opportunity to select the combination of features they prefer, given their income;

Results depend heavily on model specification;

Large amounts of data must be gathered and manipulated;

Relatively complex to implement and interpret, requiring a high degree of statistical expertise;

Time and expense to carry out an application depends on the availability and accessibility of data;

The availability of reliable price records can be a major problem.

**Related goods approaches.** A non-marketed service may be related to a marketed service. By using information about this relationship and the price of the marketed service, the analyst may be able to infer the value of the non-marketed service. This broadly defined related goods approach consists of three similar valuation techniques (UNECE 1996):

- **Barter exchange approach** – there are many forest products that are not widely traded in formal markets, for example, wild fruits, nuts and vegetables, medicines and structural fibres. However, some of these forest products may be exchanged on a non-commercial basis through a process of barter. If the bartered good that is exchanged for the forest product is also sold in a commercial market, then it may be possible to derive the value of the non-marketed good using information on the relationship (that is, the units of exchange) between the two goods and the market value of the commercial good. As with all valuation techniques, care must be taken in applying this approach. Bartering may occur in an “imperfect” non-commercial market and the rate of exchange may reflect a wider range of socio-economic factors than just the value of the goods exchanged. There are few, if any, studies that have attempted to infer the value of a forest product from the marketed value of a bartered good. However, this should not exclude the technique from being considered a potentially useful valuation approach, especially in developing countries where bartering is common.

- **Direct substitute approach** – if forest goods used directly are non-marketed, then the value of their use may be approximated by the market price of similar goods or the value of the next best alternative/substitute good. The extent to which the value of the marketed good reflects the value of the non-marketed good depends, to a large extent, on the degree of similarity or substitution between the two goods. That is, if the goods are perfect substitutes then their economic values should be very close. As the level of substitution decreases, so does the extent to which the value of a marketed good can be taken as an indication of the non-marketed forest good. Once again, market imperfections may distort the economic value of the good or service reflected in the marketplace.

- **Indirect substitute approach** – if the value of the substitute good cannot be determined directly from the market then it may be possible to derive its value indirectly, by analysing the change in value of economic output caused by a change in the use of the substitute good as an input into production. However, the indirect substitute approach is necessarily based on fairly stringent assumptions about the level of substitution between the two goods, the role of the substitute good as an input into economic output, and the value of the economic output. This technique is also fairly data-intensive. Given the tenuous link between the item being valued and the actual valuation procedure and the heavy data requirements, this approach can be expected to provide only rough indications of value.

**Reduced revenues and increased costs method** – it is based on valuing the compensation
requirements for the production of public goods, which are reflected in reduced revenues and increased costs (Sarvašová et al. 2014).

Production function-based methods - estimate how much a given ecosystem service contributes to the delivery of another service or commodity which is traded on an existing market. That means, this approach is based on the contribution of ES to the enhancement of income or productivity (Patanayak, Kramer 2001). The idea thus is that any resulting “improvements in the resource base or environmental quality” as a result of enhanced ES, “lower costs and prices and increase the quantities of marketed goods, leading to increases in consumers’ and perhaps producers’ surpluses.” (Pascual, Muradian 2010).

➨ Suitability for the FES to be valued – applicable to regulating and supporting services

➨ Benefits of the method (Pascual, Muradian 2010):

- Improvement in resource base or environmental quality, i.e. enhanced ES, lowers costs and prices or increases quantity of goods;
- Use data from actual markets, and thus reflect actual preferences or costs to individuals;
- Data are relatively easy to obtain.

➨ Limitations of the method (Daily 1997; Pascual, Muradian 2010):

- Requires knowledge of relationships between ecosystems services and valued end points;
- Adequate data on and understanding of the cause-effect linkages between the ecosystem service being valued and the marketed commodity are often lacking;
- Rarely understood well enough to quantify how much of a service is produced, or how changes in ecosystem condition or function will translate into changes in the ES delivered;
- The interconnectivity and interdependencies of ES may increase the likelihood of double-counting ES;
- The non-use benefits associated with a resource are not taken into account.

2.2.2 Cost based valuation methods

Cost based approaches can be used to infer value from the amount spent on providing a service. However, they may not be an accurate measure of value because, individuals for example, may value a service more than the cost of providing it.

Indirect opportunity costs - is used to calculate the value of non-market environmental goods when individual labour is involved in harvesting or collection. The opportunity cost method provides quick and straightforward information on the forgone development costs of preservation, but it does not provide an estimate of the social benefits of preservation. The basic assumption of this technique is that the decision to spend time in the collection and harvesting of, for example, non-wood forest product (NWFP) is weighed against alternative productive uses of labour. However, in many cases, it is almost impossible to assess how much labour is used for collecting NWFPs. Occasional NWFP collection is often a leisure/recreational pursuit. These “user cost-based techniques” suffer from the same deficiency - what something is worth has no necessary relationship to the costs involved to produce it. The fact that it is hard to estimate the users’ cost to produce, for such joint products as NWFPs in the informal sector, makes this technique somewhat dubious (UNECE 1996).

Restoration cost - uses costs of restoring ecosystem goods or services (Pascual, Muradian 2010). It is based on the idea that given an alternative land-use option the non-marketed benefits provided by an intact ecosystem or the particular services provided by such an ecosystem can be measured by estimating what it would cost to re-create the original ecosystem (or environmental service). The assumption is that by restoring the original ecosystem the original level of benefits will be restored (UNECE 1996).

In the case of primary forests, this method would involve costing the restoration of the original forest cover. Clearly, this is not something that, even with active intervention in silviculture and forest management, could be concluded quickly if it could be accomplished at all. Such considerations suggest that the technique is unlikely to prove useful (UNECE 1996).
Replacement cost method – the loss of a natural system service is evaluated in terms of what it would cost to replace that service (Sarvašová et al. 2014). It is perhaps more realistic method of re-creating non-marketed benefits consists in replacing specific natural ecosystem functions or assets with man-made production processes and capital, instead of relying on the restoration of the original ecosystem or function to provide the original level of benefits. This technique generates a value for the benefits of an environmental service by estimating the cost of replacing the benefits with an alternative service. It rests on the availability of such an alternative for the original service. The alternative should produce, as nearly as possible, the same level of benefits supplied by the resource or environmental function being valued. This technique rests heavily on the assumption that replacing the original good or service is worthwhile, and that the benefits generated by the investment in replacement outweigh the costs of replacement (UNECE 1996; FOREST EUROPE 2014).

Suitability for the FES to be valued – provisioning services (non-wood forest products, water, raw material), regulation services (water regulation, water purification and waste treatment, soil formation), cultural services (aesthetic values)
Costs incurred by individuals in order to avoid damages at already existing services can be interpreted as a lower bound of the willingness to pay for this service; 

Useful in estimating indirect use benefits when ecological data are not available for estimating damage functions with first-best methods. 

**Limitations of the method** (Pascual, Muradian 2010; FOREST EUROPE 2014): 

- The replacement cost method can undervalue the benefits as only market information is used and wider social, environmental and economic benefits are not considered; 
- No measure of individual utility if only decision-maker’s preferences count or if only “experts” decide about costs of public budgets; 
- In some cases, the replacement cost may be higher than the benefit gained from the replacement (and economic efficiency demands that the marginal benefit equals marginal cost); 
- May overstate willingness to pay if only physical indicators of benefits are available; 
- There are also data issues and limitations, and these have an important impact on the results that can be generated by this method.

**Relocation cost** - method uses the costs of relocating threatened communities (Barbier 2007; Pascual, Muradian 2010). This technique involves estimating how much it would cost to relocate (and re-equip) communities in order that they might obtain a level of benefits in their new location similar to those derived at their original site. Instead of investigating the cost of bringing substitute benefits to populations in existing sites, this technique examines the potential for moving people to alternative locations where such benefits exist. 

Application of the relocation cost technique to forests is typically restricted to a different purpose, namely, assessing the direct costs of establishing new protected areas that require the resettlement of forest-dwelling communities. 

**Benefits of the method** (Pascual, Muradian 2010): 

- Only useful in valuing environmental amenities in the face of mass dislocation such as a dam project and establishment of protected areas. 

**Limitation of the method** (Pascual, Muradian 2010): 

- Benefits provided by the new location are unlikely to match those of the original location.

**Preventive/defensive expenditure** - uses the costs of preventing damage or degradation of environmental benefits. It involves obtaining a figure for what it would cost to maintain environmental benefits by investing in the prevention of their degradation (Pascual, Muradian 2010; FOREST EUROPE 2014). 

**Suitability for the FES to be valued** - water quality, erosion protection services, water purification services, storm protection services, and nursery services. 

**Benefits of the method** (Pascual, Muradian 2010; FOREST EUROPE 2014): 

- Useful in estimating indirect use benefits when prevention technologies are available; 
- Costs incurred by individuals in order to avoid damages at already existing services can be interpreted as a lower bound of the willingness to pay for this service. 

**Limitations of the method** (Pascual, Muradian 2010; FOREST EUROPE 2014): 

- Mismatching the benefits of investment in prevention to the original level of benefits may lead to spurious estimates of willingness to pay; 
- No measure of individual utility if only decision-maker’s preferences count.

### 2.2.3 Non-monetizing valuation methods

In some cases, monetary valuation of ecosystem services is either inappropriate or not possible. This could be due to the nature of the ecosystem service, the degree of uncertainty surrounding environmental change, non-marginal changes or because of objections to monetary valuation from stakeholders and/or the researchers.
involved in the study. In this situation a variety of qualitative valuation methodologies can be undertaken. Some of these are briefly summarised below.

**Individual index-based methods**—participants are asked to value particular ES by ranking them in surveys. By looking for trends in the ranking process, the most valuable ES can be determined. Because they are not expressed in financial terms, the applicability of this method to management decisions is limited. It does, however, provide a means to determine value for services such as cultural preservation which cannot be expressed monetarily. Individual index-base methods including rating or ranking choice models (Young Ko 2007) and expert approach, that consists generally of two steps (Šišák et al. 2010):

1. In the first step, mutual relative importance of the respective non-market forest service to the market one whose pecuniary value is known is derived, based on points or percentage of importance, and a coefficient is created. The relative importance and coefficient is expressed by a set of experts;
2. In the second step, the pecuniary value of respective non-market forest service is expressed on the basis of the market service price and the derived coefficient (by multiplication of the two mentioned quantities).

**Group-based methods**—use focus groups and citizen juries rather than individual preferences to rank ES. The group-based method offers advantages similar to the individual index-based method in that it provides a mechanism to value services that cannot be expressed monetarily. The main drawbacks to this method are representativeness and the fact that sociological factors such as peer pressure can influence how participants rate (Young Ko 2007). Group-based methods comprise (Frederick et al. 1996; Aldred, Jacobs 2000; Gregory, Wellman 2001; Howarth, Wilson 2006):

- **Voting mechanisms** is used widely to choose options, often because of its ease of application. It is used most often in informal settings to generate group preferences.
- **Focus groups** is used to elicit information about values and preferences from small groups of relevant members of the public engaging in group discussion led by a facilitator. However, the use of qualitative measures and the uncertainty of any generalizations of results from small respondent samples limit the utility of these methods for formal policy and decision making.

- **Citizen juries** also incorporate elements of the deliberative valuation process. In principle, a jury could be asked to generate a value for how much the public would, or should, be willing to pay for a possible environmental improvement, or, conversely, willing to accept for an environmental degradation. In contrast to estimates of willingness to pay derived from economic valuation methods, the estimates from citizen juries would not reflect the budget constraints of the individual participants and would reflect community based values rather than economic values. To the extent that a citizen jury engages in group deliberation, resulting value estimates also would reflect constructed values.

- **Stakeholder analysis** is a process of identifying stakeholders and categorising them according to their relationship with the issue or activity and determining what this means in terms of how / whether to work with them. Stakeholder analysis helps the manager to identify each stakeholder, each stakeholder’s interest, and the changes in stakeholder perceptions of issues and in the balance of influence over time.

**Measures of attitudes, preferences and intentions**—are social-psychological approaches to assess the value of ecosystems and ES employ a number of methods to identify, characterize, and measure the values people hold, express, and advocate with respect to changes in ecological states or their personal and social consequences. These methods elicit value-relevant perceptions and judgments, typically expressed as choices, rankings, or ratings among presented sets of alternative ecosystems protection policies and may include comparisons with potentially competing social and economic goals. Individuals making these judgments may respond on their own behalf or on behalf of others. The basis for judgments can
be changes in individual well-being or in civic, ethical, or moral obligations (USEPA SAB 2009).

Social-psychological value-assessment approaches have relied most strongly on survey methods. Survey questions eliciting information about attitudes, preferences, and intentions are most often presented in a verbal format, either in face-to-face or telephone interviews or in printed questionnaires. Assessments of values for ecosystems and ES can be well-conveyed in perceptual surveys and conjoint surveys. Quantitative analyses of survey responses are usually interpreted as ordinal rankings or rough interval-scale measures of differences in assessed values for the alternatives offered. Survey questions about social and psychological constructs may be especially useful when the values at issue are difficult to express or conceive in monetary terms, or where monetary expressions are likely to be viewed as ethically inappropriate (USEPA SAB 2009).

**Civic valuation** - Civic valuation seeks to measure the values that people place on changes in ecosystems or ES when explicitly considering or acting in their role as citizens. These valuation methods often seek to value changes that would benefit or harm the community at large. They purposefully seek to assess the full value that groups attach to any increase in community well-being attributable to changes in the relevant ecosystems and services (USEPA SAB 2009).

Civic valuation, like economic valuation, can elicit information about values either through revealed behaviour or through stated valuations. One source of information based on revealed behaviour is votes on public referenda and initiatives involving the provision of environmental goods and services. Another source is community decisions to accept compensation for permitting environmental damage. Where revealed values are difficult or impossible to obtain, citizen valuation juries or other representative groups can be charged with determining the value they would place on changes in particular ecological systems or services when acting on behalf of, or as a representative of, the citizens of the relevant community (USEPA SAB 2009).

**Decision science approaches** - derive information about people's values through a deliberative process that helps individuals to understand and assess trade-offs among multiple attributes. The ultimate goal is for an individual or group to assign scores to alternatives (e.g., different projects) that can then be used to choose among those alternatives, recognizing that those alternatives will differ along a number of relevant dimensions or attributes. Generally, one alternative will score higher along some dimensions but not others, suggesting that trade-offs must be made when choosing among alternatives (Clemen 1996; Arvai, Gregory 2003).

- **Suitability for the FES to be valued** - spiritual, cultural services (Kenter et al. 2015)
- **Benefits of the method** (Arvai, Gregory 2003; USEPA SAB 2009):
  - Ability to not only integrate multiple attributes value, but also engage a broad spectrum of stakeholders, holders of traditional ecological or cultural knowledge, and technical experts in the valuation process;
  - High potential for identifying changes in ecosystems and their services that are likely to be of greatest concern to people;
  - Method may potentially overcome (primarily) public or stakeholder objections to other approaches that are not perceived to adequately include moral and other non-monetary aspects of value.
- **Limitations of the method** (Arvai, Gregory 2003; USEPA SAB 2009):
  - The trade-offs are typically not easy to make;
  - It requires time and expertise resources;
  - Engaging with stakeholders and technical experts to identify attributes that will be the focus of analysis, collecting data that characterizes these attributes, and the process of making trade-offs all will require effort on the part of agencies on environmental protection.

**Ecosystem benefit indicators** - offer quantitative metrics that are generally correlated with ecological contributions to human well-being and hence can serve as indicators for these contributions in a specific
setting. They use geo-spatial data to provide information related to the demand for, supply (or scarcity) of, and complements to particular ES across a given landscape, based on social and biophysical features that influence – positively or negatively – the contributions of ES to human well-being (Boyd et al. 2001; Wainger et al. 2001; Boyd, Wainger 2002; Boyd 2004).

Ecological benefit indicators can serve as important quantitative inputs to valuation methods as diverse as citizen juries and economic valuation methods. They provide a way to illustrate factors influencing ecological contributions to human welfare in a specific setting. The method can be applied to any ecosystem service where the spatial delivery of services is related to the social landscape in which the service is enjoyed. However, although the resulting indicators can be correlated with other value measures, such as economic values, they do not themselves provide measures of value (Boyd et al. 2001; Wainger et al. 2001; Boyd, Wainger 2002; Boyd 2004).

Suitability for the FES to be valued – can be applied to any ecosystem service where benefits are related to the spatial delivery of services and social landscape in which the benefit is enjoyed, except existence benefits.

Benefits of the method (USEPA SAB 2009):
- Relatively non-technical way to express the factors that contribute to conventional economic measures of benefits provided by ES;
- Simple and transparent;
- Can be used to communicate and educate.

Limitations of the method (USEPA SAB 2009):
- Do not directly yield money-based ecological benefit estimates;
- Do not in themselves weight or estimate the trade-offs associated with different factors relating to benefits;
- Because indicators can be cheaper to generate than econometric value estimates they better allow for landscape assessment of multiple services at large scales.

Biophysical ranking methods – try to value ES’ values based on the quantification of biophysical indicators. Quantification of ecological changes in biophysical terms allows these changes to be ranked based on individual or aggregate indicators for use in evaluating policy options based on biophysical criteria previously determined to be relevant to human well-being. Possible indicators include measures of biodiversity, biomass production, carbon sequestration or energy and material use (Stoms 2005; USEPA SAB 2009).

Use of a biophysical ranking does not explicitly incorporate human preferences. Rather, it reflects either a non-anthropocentric theory of value (based, for example, on energy flows) or a presumption that the indicators provide a proxy for human value or social preference. This latter presumption is predicated on the belief that the healthy functioning and sustainability of ecosystems is fundamentally important to the well-being of human societies and all living things, and that the contributions to human well-being of any change in ecosystems can be assessed in terms of the calculated effects on ecosystems. Opinion is mixed – among both committee members and the broader scholarly community – on whether it is an asset or a drawback that these ranking methods are not tied directly to human preferences. Another problem is measuring trade-offs between services which are measured in different biophysical units (Stoms 2005; USEPA SAB 2009).
3. Conclusions and Recommendations

The review of valuation methods was devoted especially to methods for economic valuation of ecosystem services. For the sake of the complexity of the analysis also methods of ecological evaluation of ecosystem services were touched as well.

3.1 Selecting the Economic Valuation Method

Estimating the value of the various services and benefits that ecosystems generate may be done with a variety of valuation approaches and methods. This diversity is an inevitable consequence of the diversity of FES and the conditions under which they are provided as well as different types of valuation questions addressed. Some of the valuation techniques are broadly applicable, some are applicable to specific cases, and some are tailored to particular data sources.

While market price methods as well as revealed and stated preference methods are theory-driven and consistent with economic welfare theory, cost-based approaches rely on empirical and pragmatic considerations.

Several issues should be considered when selecting valuation methods to be applied. The general recommendation is, first, that it must fit to the given valuation problem (e.g. services to be valued, valuation context and geographical scope); and second, that it should be manageable with available resources (in terms of data, time, analysis skills, and other requirements). However, the final selection of a valuation method is also determined by a number of other factors and conditions and should be consulted with experienced economists.

A point that should be discussed is the reliability of valuation methods and sometimes high uncertainty implicit in the results of these methods. There could be the case that different valuation methods provide divergent results. It is, hence, difficult to compare results which are produced by different methods.

The challenge is also appropriate application of valuation methods to particular ecosystem services. Different ecosystem services require different valuation methods. Provisioning services usually produce goods that are physically tradable, such as timber, therefore the market based methods are most appropriate to measure their values. Examples of market based methods are the use of direct market prices, production function methods, calculation of replacement costs, defensive expenditures etc.

As many ecosystem services (e.g. regulating services) are not usually traded in markets and their values are not indicated by market prices, it is necessary to assess the relative economic worth of these services using non-market valuation techniques.

The values of cultural ecosystem services only emerge through their (subjective) effect on the wellbeing of people, such as through the perceived value of recreation, and are best measured with revealed and stated preference methods.

Economic valuation of forest ecosystem services usually relies on the notion of
<table>
<thead>
<tr>
<th>Classification of ES according to TEEB (2010)</th>
<th>Suitable valuation method</th>
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<tbody>
<tr>
<td><strong>PROVISIONING SERVICES</strong></td>
<td></td>
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<tr>
<td>Raw materials</td>
<td>Market prices, Contingent valuation method; Choice modelling, Value transfer method; Market price method; Efficiency (shadow) prices; Restoration costs method; Replacement costs method</td>
</tr>
<tr>
<td>Food / Raw materials</td>
<td>Market prices, Contingent valuation method; Choice modelling, Value transfer method; Market price method; Efficiency (shadow) prices; Restoration costs method; Replacement costs method</td>
</tr>
<tr>
<td>Water supply</td>
<td>Market, prices, Contingent valuation method; Choice modelling, Value transfer method; Related goods approaches; Indirect opportunity costs; Restoration costs method; Replacement costs method</td>
</tr>
<tr>
<td>Genetic resources</td>
<td>Contingent valuation method; Choice modelling, Value transfer method; Related goods approaches; Indirect opportunity costs</td>
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<tr>
<td><strong>REGULATING SERVICES</strong></td>
<td></td>
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<tr>
<td>Biological control</td>
<td>Contingent valuation method; Choice modelling, Averting behavioural method; Value transfer method; Related goods approaches; Production function-based methods; Indirect opportunity costs</td>
</tr>
<tr>
<td>Regulation of water flows</td>
<td>Contingent valuation method; Choice modelling, Averting behavioural method; Value transfer method; Related goods approaches; Production function-based methods; Indirect opportunity costs; Restoration costs method</td>
</tr>
<tr>
<td>Disturbance prevention or moderation</td>
<td>Contingent valuation method; Choice modelling, Averting behavioural method; Value transfer method; Related goods approaches; Production function-based methods; Indirect opportunity costs; Replacement cost method; Preventive/defensive expenditures</td>
</tr>
<tr>
<td>Waste treatment (water purification)</td>
<td>Contingent valuation method; Choice modelling, Averting behavioural method; Value transfer method; Hedonic pricing method; Related goods approaches; Production function-based methods; Indirect opportunity costs; Restoration costs method; Replacement costs method; Preventive/defensive expenditures</td>
</tr>
<tr>
<td>Air purification</td>
<td>Contingent valuation method; Choice modelling, Averting behavioural method; Value transfer method; Hedonic pricing method; Related goods approaches; Production function-based methods; Indirect opportunity costs</td>
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<tr>
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<td>Suitable valuation method</td>
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<tr>
<td><strong>REGULATING SERVICES</strong></td>
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<tr>
<td>Climate regulation (incl. C sequestration)</td>
<td>Contingent valuation method; Choice modelling; Averting behavioural method; Value transfer method; Related goods approaches; Production function-based methods; Indirect opportunity costs</td>
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<tr>
<td>Erosion prevention</td>
<td>Contingent valuation method; Choice modelling; Averting behavioural method; Value transfer method; Related goods approaches; Production function-based methods; Indirect opportunity costs; Replacement costs method; Preventive/defensive expenditures</td>
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<tr>
<td>Maintaining soil fertility</td>
<td>Contingent valuation method; Choice modelling; Averting behavioural method; Value transfer method; Related goods approaches; Production function-based methods; Indirect opportunity costs; Restoration costs method</td>
</tr>
<tr>
<td>Pollination</td>
<td>Contingent valuation method; Choice modelling; Averting behavioural method; Value transfer method; Related goods approaches; Production function-based methods; Indirect opportunity costs</td>
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<tr>
<td><strong>HABITAT SERVICES</strong></td>
<td></td>
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<tr>
<td>Maintenance of genetic diversity (especially in gene pool protection)</td>
<td>Contingent valuation method; Choice modelling; Value transfer method; Related goods approaches; Indirect opportunity costs</td>
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<tr>
<td>Lifecycle maintenance</td>
<td>Contingent valuation method; Choice modelling; Value transfer method; Related goods approaches; Indirect opportunity costs</td>
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<tr>
<td><strong>CULTURAL &amp; AMENITY SERVICES</strong></td>
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<td>Spiritual experience and sense of place</td>
<td>Contingent valuation method; Choice modelling; Value transfer method; Related goods approaches; Indirect opportunity costs; Decision science approach</td>
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<tr>
<td>Inspiration for culture, art and design</td>
<td>Contingent valuation method; Choice modelling; Value transfer method; Indirect opportunity costs</td>
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<tr>
<td>Recreation and tourism</td>
<td>Contingent valuation method; Choice modelling; Value transfer method; Travel costs method; Hedonic pricing method; Related goods approaches; Indirect opportunity costs; Random utility/discrete choice</td>
</tr>
<tr>
<td>Aesthetic information</td>
<td>Contingent valuation method; Choice modelling; Value transfer method; Hedonic pricing method; Related goods approaches; Indirect opportunity costs; Restoration costs method; Replacement costs method</td>
</tr>
<tr>
<td>Information for cognitive development</td>
<td>Contingent valuation method; Choice modelling; Value transfer method; Hedonic pricing method; Related goods approaches; Indirect opportunity costs</td>
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consumers’ willingness to pay. Some of the respective methods rely on the revealed behaviour of the users for these services (i.e. revealed preference approaches), while others use surveys and directly ask users about their willingness to pay for certain services (i.e. stated preference approaches).

The advantage of revealed preference approach is that they are based on actually observed behaviour. The value of forest services in question can be either derived directly (e.g. from market prices) or indirectly from surrogate markets that have a relationship with the forest service of interest. However, the applicability of these methods is limited only to a few forest ecosystem services (e.g. recreation, tourism and amenities). The most used revealed preference method is the hedonic pricing method and the travel cost method.

Stated preference methods are based on hypothetical rather than actual behaviour data. The value of a forest service is derived from people’s responses to questions/choices describing hypothetical markets or situations. The methods in this group can be applied to all types of forest ecosystem services and allow to estimate both use and non-use values. However, their main disadvantages are that they are based on hypothetical situations (no real market transaction is performed and the received answer might not reflect the real situation). Methods can be limited by the ability of respondents to understand the nature of the service. Respondents in a valuation exercise may well not appreciate the impact that an ecosystem service might have on their wellbeing. As it is well known in the valuation literature, large impacts can result from the hypothetical nature of valuation questions as respondents may overestimate what they would be prepared to pay where they do not actually have to make a payment. The most used stated preference methods include the contingent valuation method and choice modelling techniques.

The table below provides a list of ecosystem services classified according to TEEB (2010) and methods suitable for their valuation. The table in an interactive form including various case examples on valuation of FES is available at the FOREST EUROPE Web-based Portal on Forest Ecosystem Services https://foresteurope.org/valuation-forest-ecosystem-services/ (for details see Part II of this publication).

Case examples of valuation methods are provided especially for the contingent valuation method, choice modelling, the value transfer method, the market price method, the travel cost method, and the hedonic pricing method. This is because other respective methods are only rarely used (most probably because they are not practically applicable), or because there is an overlap with another method (some valuation methods have several variants, and/or sub-categories, which are in fact almost identical, and are being used for essentially the same purpose).

3.2 Limitations to Economic Valuation

During the past three decades, the economic valuation approaches have improved considerably, however some limitations still exist. These limitations can be summarised as follows (Turner et al. 2003; DEFRA 2007; European Commission 2008; TEEB 2008; Thorsen et al. 2014):

**Interdependence of ecosystems and their services.**
This includes both the interdependence within an ecosystem (i.e. various components of an ecosystem interact to provide a certain service) and interdependence between ecosystems (i.e. various ecosystems may interact to provide a certain service). For valuation, this means that the economic value of any one service may depend on its relationship with other services, and therefore an assessment of the value of one service may not easily take into account how other services are being affected.

**Marginality.**
The economic valuation is meaningful when considering small, marginal changes in the provision of ecosystem services.

**Double counting.**
Some ecosystem services are not complementary, the provision of one is precluded by others (trade-offs). Therefore, to prevent double-counting, the full range of complementary and competitive services must be distinguished before any aggregation of values is completed.

**Spatial issues.**
Ecosystem functions and their capacity to supply services to a particular human population are often best evaluated across their full geographical extent, which may not fit well with the spatial scale of valuation context. The
valuation should take into account the complete population affected, whose values may be affected by the changes in ecosystem services supply. To estimate appropriate values, it is necessary to understand whether an ecosystem service is local, regional, national or global.  

**Temporal issues.**  
Impacts on ecosystems and their services may extend well beyond a standard time period of a given policy (project) appraisal. It is therefore important to account for any temporal distribution of costs and benefits. This is normally done by discounting, using an appropriate discount rate, which converts all costs and benefits to present values so that they can be compared. However, the choice of the discount rate usually requires additional assumptions. In the Green Book (2003) the British Treasury guidelines recommend using different (declining) discount rates over the longer term. The reason for this is that uncertainty increases as we look further into the future. The choice of discount rate can make a very significant difference in terms of the final outcomes of any cost-benefit analysis. The timeframes over which costs and benefits are considered should depend on the duration over which the costs and benefits will be realised.

**Environmental limits.**  
The services that ecosystems provide depend not only on the scale and function of the ecosystem but also, crucially, on its conditions and biodiversity levels. As the state of an ecosystem deteriorates, the services it provides are deemed to diminish. Sometimes, this may be a gradual process, but in other circumstances a threshold may be reached. Beyond this threshold, an irreversible change in the ecosystem may occur (e.g. total collapse), resulting in permanent loss of services.  

An economic valuation study typically estimates values only for a marginal change in a service or ecosystem condition at a few points along the demand curve. Applying these marginal values to non-marginal changes in ecosystems is therefore not appropriate.  

**Dealing with uncertainty.**  
There is considerable uncertainty surrounding both the knowledge regarding functional aspects of ecosystems and valuation of ecosystem services. Even among specialist scientific communities, there is no consensus on certain aspects, for example, what services are provided by different ecosystems, how these may change over time and how changes in ecosystems may affect the quantity and quality of the services they provide. This is further complicated by the fact that ecosystems may not respond to change in a linear fashion; there may be thresholds beyond which an ecosystem responds in a previously unknown manner. Under such circumstances, consideration needs to be given to the uncertain future losses or gains that might be associated with potential change. One option for accounting for uncertainty is to conduct a sensitivity analysis by identifying areas of uncertainty and testing how sensitive the evaluation outcomes are to changes in values or assumptions used in valuing ecosystem services.  

**Data transfer and knowledge gap.**  
The quality and availability of valuation data could be improved by exploiting knowledge from valuation studies from other locations. However, data transfer from other studies is challenging. Issues are related to the need for good quality studies of similar situations, their social and environmental context, to changing characteristics in different time periods and the inability to deal with the valuation of novel impacts. Thus, an effort on collecting existing studies and/or improving open access to existing databases of valuation studies with specific focus on European conditions should be made. This would enable to recognize what type of studies and/or data is available. An example is a database on Forest Ecosystem Service Valuation Studies in France and in the German-speaking countries (Elsasser et al., 2009; Elsasser et al., 2016). The database is a first step towards facilitating the access to studies about forest externalities for researchers all over the world. Another example is Woodland Valuation Tool that enables those involved in forestry management to search for, and cross-reference methods and scenarios associated with different trees and woodland to test out their potential benefits and pitfalls at the planning stage. There are also other databases such as EVRI or Envalue, however, they contain rather general information about valuation studies and do not...

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5) https://forestry.gov.scot/sustainable-forestry/economic-research/non-market-values
cover methodological details and/or are not generally accessible, which is not sufficient for data transfer.

The following points and caveats are important to note in case of using valuation approaches (European Commission 2008; Standing Forestry Committee 2008; Thorsen et al. 2014; FOREST EUROPE 2014):

- Methods and their results are based on theoretical background, purpose of valuation, socioeconomic conditions, and data availability.

- The role of valuation is to show the contribution of ecosystem services to the wellbeing of people, to increase awareness of existing benefits as well as creating sense of ownership and commitment among stakeholders. However, valuations themselves do not determine whether a service should go to market (let alone the questions of who should pay and how much he or she should pay); for that, negotiations between providers and beneficiaries are often necessary.

- One of the main limitations of economic valuation is that the resulting estimates are often highly context dependent, being sensitive to both the methods selected and assumptions used. For example, some methods mainly focus on marketed services, but omit non-market values. In addition, the selected ecosystem service, valuation period and discount rate have profound effects on the estimates.

- Values estimated in different contexts should not be compared directly. One of the limitations of valuation methods is that, in general, they do not allow direct comparison of economic values estimated in different studies, or the use of the estimated values to express the relative economic importance of different forest goods and services. These limitations result from differences in valuation objectives, methods applied, data accuracy, considered target populations, value units, etc.

- There are no generally accepted procedural rules for monetary valuation of forest ecosystem services which would
allow for a simple “cookbook approach”.

- However, there are good technical guidance (e.g. guidance developed by DEFRA\(^6\), COST Action E45 EUROFOREX\(^7\)), which can help to decide how to implement valuation and how to deliver such values, including the vital step of identifying the beneficiaries of FES and, therefore, potential demand for them.

- Valuation is one of the element of a more complex decision making process. However, it can broaden the perspective and information base for a better informed policy making.

- From the view of implementation of FES for policy support and consulting, successful valuation approaches and results should particularly consider rational relationships between economic, ecological and social aspects of FES provision.

Various practical barriers are reported for broader consideration regarding approaches and results of monetary valuation of services in policy decisions. They include (FOREST EUROPE 2014; Kill 2014):

1. cultural barriers - considering economic approaches for solving environmental problems is generally seen with some reservations in several European countries. Hence there is less of experience with economic valuation of environmental services in these countries (apparently there are fewer economic valuations of FES for example in the German speaking countries than in the UK or in Scandinavia);

2. methodological barriers - no generally accepted procedural rules amidst methodological complexities of valuation; and

3. political barriers - it can be much easier to communicate political decisions based on “real money” than on what some see as intangible and nebulous values based on the consumer surplus concept.

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\(^7\) [http://www.efi.int/portal/projects/cost.e45](http://www.efi.int/portal/projects/cost.e45)
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PART II. Payments for Forest Ecosystem Services
1. **Introduction to Payments for Forest Ecosystem Services**

The ecosystem services approach describes ecosystems as natural capital stocks that provide diverse goods and services for human societies (Costanza and Daly 1992; Daily 1997; de Groot et al. 2002; MA 2005; TEEB 2010; Maes et al. 2013, 2014, 2018). Despite the long-term awareness of the importance of ecological functions for livelihoods, the origins of the modern ecosystem service approach are to be found in the late 1960s and the 1970s (Ehrlich and Ehrlich 1981; Helliwell 1969; King 1966; Odum and Odum 1972).

The expansion of pricing mechanisms to ecosystem services has followed two main approaches. The first consists of a “Pigovian solution” where public intervention plays the leading role in the correction of “market failures” through state taxes and subsidies. The second approach, prominent since the 1980 and 1990s, follows a “Coasean solution” whereby correction of market failures is addressed through private transactions, often in markets where ecosystem services can be freely sold and bought. Coase (1960) identified well-established property rights and low transaction costs as necessary preconditions for efficiently addressing externalities through direct negotiation. These approaches for correcting market failures could be implemented via two main mechanisms: (i) markets for ecosystem services and (ii) payments for ecosystem services (PES).

PES schemes are most likely to emerge in situations where:

1. specific land or resource management actions have the potential to increase the supply of a particular service (or services);
2. there is a clear demand for the service(s) in question, and its provision is financially valuable to one or more potential buyers;
3. it is clear whose actions have the capacity to increase supply (for example, certain land or resource managers may be in a position to enhance supply).

Especially in the absence of a legislative framework or functioning local governance, PES offer a promising mechanism for increasing the supply and compensating the cost of provision of ecosystem services (Schomers and Matzdorf 2013) at least if there is some minimum protection of property rights. PES schemes represent a tool for maintaining and improving non-marketed ecosystem services and biodiversity levels and at the same time assuring the provision of marketed ecosystem services; and a possible path for diversifying and scaling-up various sources of funding for forestry activities. PES can create incentives for the providers of forest ecosystem services for managing forest following a multifunctional approach and keeping constant or increasing the supply of services without any loss. Forests can be managed in sustainable way, conserving the biodiversity and developing the multifunctionality of forest stands (Viszlay et al. 2016).

Payment mechanisms for many non-market forest services are mainly economic instruments which aim to internalise environmental or depletion cost through financial incentives. It should enable forest service providers, e.g. forest owners, to be able to manage sustainably the forest without incurring costs out of proportion to the personal benefit received but does not necessarily mean
that payment to the full value of all forest goods and services is possible or desirable (SFC 2008).

According to Smith et al. (2013), the term PES is used to describe schemes in which the beneficiaries, or users, of ecosystem services provide payment to the stewards, or providers, of ecosystem services. In practice, PES often involves a series of payments to land or other natural resource managers in return for a guaranteed flow of ecosystem services or, more commonly, for management actions likely to enhance their provision over-and-above what would otherwise be provided in the absence of payment. Payments are made by the beneficiaries of the services in question, for example, individuals, communities, businesses or government acting on behalf of various parties. It is argued that payments for ecosystem services can be more economically efficient and more environmentally effective (i.e. sustainable) than other incentive based approaches because they create a direct relationship between the supplier and the buyer of the service (Engel et al. 2008).

The basic idea behind PES is that those who maintain ecosystems in good condition and provide ecosystem services and incur an extra cost, should be paid for doing so. PES therefore provides an opportunity to put a price on previously un-priced ecosystem services. The novelty of PES arises from its focus on the “beneficiary pays principle”, as opposed to the “polluter pays principle” (Smith et al. 2013). PES certainly contribute to the understanding of the importance of ecosystems in good condition and their services, bringing ecological awareness, as well as active social participation in governance (FAO 2010).

Vatn (2010) states that it may be important to differentiate between “the wider concept of PES and the narrower concept of markets for environmental services”. That is also acknowledged by Wunder et al. (2008).

In the literature, the most widely accepted PES definition is according to Wunder (2005, 2007) who characterized PES by five criteria: (1) voluntary transaction where (2) a well-defined ecosystem service (or a land-use likely to secure that service) (3) is “bought” by (a minimum of one) ecosystem service buyer (4) from (a minimum of one) ecosystem service provider (5) if and only if the ecosystem service provider secures ecosystem service provision (conditionality).

A revised PES concept according to Wunder (2015) defines PES as voluntary transactions between service users and service providers that are conditional on agreed rules of natural resource management for generating offsite services.

Recently, there has been an increasing use of a less strict definition according to Mudrian et al. (2010) that relax some of Wunder’s criteria: PES are a transfer of resources between social actors, which aims to create incentives to align individual and/or collective land use decisions with the social interest in the management of natural resources. This definition is broader using incentives instead of payments, no hint at conditionality.

The definition provided by Wunder is conceptually based on the Coase theorem (Engel et al. 2008; Gómez-Baggethun et al. 2010; Muradian et al. 2010) by seeking to internalize the positive externalities that ES provide through bargaining solutions between those who provide the services and those who gain from it and is theoretically rooted in the field of environmental economics (Gómez-Baggethun et al. 2010).

Drawing on this definition, there are five key principles, which should ideally underpin any PES scheme.

1. **Voluntariness.** The voluntariness is the degree to which the contracting parties, service provider(s) and the beneficiary(ies) enter into an agreement and participate through a free and informed process of negotiation (Wunder 2005). The voluntariness principle is, therefore, a characteristic that differentiates PES from the more “government based” command and control measures. In fact, PES are negotiation processes where two or more involved parties participate with different degrees of power and participation (de Groot and Hermans 2009). Therefore, we can distinguish different degrees of voluntariness as for the actual degree of participation and level of information between the contracting parties (Fung 2006). Moreover, the role of governments and regulations may influence the voluntariness only from the supply side or the demand side, or both supply and demand side. The degree of voluntariness also relates to the concept of “additionality” (see the point No. 4).
2. **A well-defined ES.** What is bought needs to be well defined – it can be a directly measurable service or land-use caps that are likely to help providing that service.

3. **Beneficiary pays.** Payments are made by the beneficiaries of ecosystem services (individuals, communities and businesses or governments acting on behalf of various parties) (Smith et al. 2013).

4. **Direct payments.** Payments are made directly to ecosystem service providers (in practice, often via an intermediary or broker) (Smith et al. 2013). The less direct case is when governments play an intermediary role in the transaction between the final user and the service providers (Nisbet et al. 2011). The more direct case is related to where PES contracts are signed directly between beneficiaries and service providers, i.e. “bilateral agreements” (Bluemling and Horstkoetter 2007).

5. **Conditionality.** Payments are dependent on the delivery of ecosystem service benefits. In practice, payments are more often based on the implementation of management practices which the contracting parties agree are likely to give rise to these benefits (Smith et al. 2013). Conditionality is the degree to which the service provision is conditional to the payment, and vice versa.

While these principles should inform about the development of PES, in practice schemes may adhere to them to a greater or lesser degree. The literature on PES suggests that few existing schemes fulfil all these principles in practice and, as such, aiming for a “perfect” PES scheme may create unrealistic expectations (Smith et al. 2013).

Definitions of PES mentioned above stress the point of the transaction mechanisms, however Muradian et al. (2010) focuses on type of actors and outcomes of PES mechanisms, giving a broader and more comprehensive definition that better fits existing examples. Most PES are thus best described as “PES-like” or “quasi-PES” schemes implemented by public entities, often acting in a rather complex institutional framework (Leonardi et al. 2016; Bennett et al. 2014; Vatn 2010) (Figure 6). Moreover, some PES policies were initiated before the term “payments for ecosystem services” came into common usage and yet are based on the same theory.

- **PES core schemes** – only schemes that strictly follow the seven main criteria - voluntary transaction between a minimum one buyer and minimum one seller of a well-defined ES and with a strong conditionality attached.
- **PES-like schemes** – incentives comply with only some of the seven requirements. For example, some programmes may not have buyers paying voluntarily for the service or other programmes may only have a low conditionality implemented or have a weak additionally.
- **Other economic incentives** – a range of economic incentives as a PES where payments are made to achieve higher levels of ES streams in different contexts.

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**Figure 6 – PES definitions – between hard core and periphery (Wunder 2005)**
2. **Policy Drivers Underpinning Financing Mechanisms and Payments for PES in the Pan-European Region**

**Forest Europe - Ministerial Conferences on the Protection of Forests in Europe**

At the 7th [Ministerial Conference on the Protection of Forests in Europe](#) held in 2015 in Madrid, FOREST EUROPE signatories expressed their commitment to recognise the key role of forest ecosystem services (FES). In the [Madrid Ministerial Resolution 1](#) "Forest sector in the center of a green economy", signatory countries committed themselves inter alia to:

- **Promote the exchange of information on methodologies and practices on the valuation of and payments for forest ecosystem services as well as policy approaches.**
- **Make further efforts to have the full value of forest ecosystem services better reflected in forest related policies and tools inter alia national forest programmes or equivalents, guidelines, market based instruments and payments for ecosystem services.**

To follow-up these commitments a FOREST EUROPE Expert Group was established with the aim to give recommendations to policy makers on the pan-European approaches to valuation of forest ecosystem services and means to facilitate its implementation, bearing in mind that there can be a range of different possibilities. The report of the Expert Group identifies the challenges for valuing forest ecosystem services and further implementation of PES. In the Expert Group report, PES are identified as a key market-based instrument.

Prior to the Madrid and Oslo commitments, water related services were addressed in [Warsaw Declaration](#) (2007) and [Warsaw Resolution 2](#) "Forest and Water" (2007) signed at the [5th Ministerial Conference](#) held in Warsaw in 2007. In the Warsaw Resolution, signatories committed themselves to implement tools for securing water-related services provided by forests, such as payments for ecosystem services.
Essential benefits provided by forests and the importance of the value of forest goods and services are recognised in the Vienna Living Forest Summit Declaration “European Forests – Common Benefits, Shared responsibilities” (2003) signed at the 4th Ministerial Conference on the Protection of Forests in Europe, held in Vienna in 2003. The signatory countries committed to promote incentives that have positive impacts on sustainable forestry, and also to the removal of incentives that have negative impacts.

The outcomes of Ministerial Conferences quoted here have had a strong influence on the forest policy development within the FOREST EUROPE signatory countries. An example is the EU Forest Strategy.

A new EU Forest Strategy: for Forests and the Forest-Based Sector

In 2013, the Commission adopted a new EU Forest Strategy (COM(2013) 659 final) which gives a new framework in response to the increasing demands put on forests and to significant societal and political changes that have affected forests over the last 15 years. Its main aim is ensuring the sustainable forest management and the multifunctional role of forests, delivering multiple goods and services in a balanced way and ensuring forest protection. Specifically, the Multi-Annual Implementation Plan (SWD(2015) 164 final) of the new EU Forest Strategy indicates PES as an innovative mechanism fostered by Member States and the Commission to finance the maintenance and restoration of ecosystem services provided by multifunctional forests.

Innovating for Sustainable Growth: A Bioeconomy for Europe

The EU's Bioeconomy Strategy and its Action Plan adopted in 2012 aim to pave the way to a more innovative, resource efficient and competitive society that reconciles food security with the sustainable use of renewable resources for industrial purposes, while ensuring environmental protection. The Bioeconomy Strategy supports the implementation of ecosystem-based management with the scope of look for synergies and complementarities
with environmental policies, sustainable use of natural resources, protection of biodiversity and habitats, as well as provision of ecosystem services. Within this approach PES are seen as valuable instruments for supporting the provision of non-marketed forest services and biodiversity protection.

**Our Life Insurance, our Natural Capital: an EU Biodiversity Strategy to 2020**

This strategy is aimed at reversing biodiversity loss and speeding up the EU’s transition towards a resource efficient and green economy. It is an integral part of the Europe 2020 Strategy, and in particular the resource efficient Europe flagship initiative. The most important actions under this strategy in connection with ecosystem services and PES are Action 5: Improve knowledge of ecosystems and their services in the EU, and Action 11: Encourage forest holders to protect and enhance forest biodiversity. Specifically, Action 11 suggests fostering innovative mechanisms, including PES, to underpin the maintenance and restoration of ecosystem services provided by multifunctional forests.

**EU Action Plan for Nature, People and the Economy**

The Action Plan was adopted by the European Commission in 2017 to reach the EU’s Biodiversity targets for 2020. The Action Plan focuses on four priority areas and comprises 15 actions to be carried out between now and 2019, whereas Action 1 aims to promote the integration of ecosystem services into decision-making.
3. **Actors Involved in Financial Mechanisms and Payments for PES**

A PES scheme often involves different “stakeholders”, a term used here covers all those with an economic or socio-cultural interest (whether expressed or not) in the ecosystem services provided by a PES scheme. The term encompasses anyone with a “stake” in an issue, both those actors with influence (i.e. those controlling a resource), and those influenced (i.e. those affected by a change in the resource). Active involvement of stakeholders is here termed “participation”, which may range from “consultation” (effective or not) through to more or less complete control of scheme decision making (Thomson et al. 2014).

Although PES theory mainly refers to two groups of actors (service providers and service buyers), other actors can influence the design and implementation of the contractual agreement. We can therefore summarize the main groups that are typically involved in a PES scheme as follows:

- **Buyers** are those who are willing to pay for an improved or safeguarded or restored ecosystem service. In deciding whether to pay, the service buyers have to evaluate if it is worth paying for or if some other alternatives could be economically better for them (Smith et al. 2006). These include citizens, water utilities, municipalities, beverage companies, etc.

- **Sellers or service providers** are land and/or forest managers whose change of management practice can potentially secure or improve supply of the ecosystem service. The service sellers provide the effort, change their habits, ways of working or make the concessions. They have to evaluate if it is economically worthwhile for them to make this change, if it will decrease profits. If they behave economically, they will not accept less than the opportunity cost of the scheme. This is the profit they would lose as a consequence of the changes in land use and management or restrictions on land use needed to comply with the scheme. Payment must be at least equal to the foregone net profit of upstream service sellers (Smith et al. 2006).

- **Knowledge providers** include resource management experts, valuation specialists, land use planners, universities, participation experts, business and legal advisors who can provide knowledge essential to scheme development.

- **Regulators** can influence the institutional framework for PES establishment positively or negatively. On the one hand, regulators’ influences can be conducive e.g. if they secure relevant property rights and/or facilitate the start-up and the effectiveness of PES mechanisms. On the other hand, existing command and control measures may also interfere with new rules necessary for PES establishment.
Donors - funding agencies or sponsors are public or private entities that fund the start-up and/or feasibility studies for a PES scheme. Intermediaries can serve as agents linking buyers and sellers and can help with scheme design, administration, implementation and operation. The role of intermediaries varies depending on the context, but they may provide information, additional funding, act as brokers, help build trust and facilitate transactions between the PES parties (Schomers and Matzdorf 2013), and reduce the overall costs of a PES initiative (Vatn 2010; Kemkes et al. 2010). They can come from the public, civil, private, or academic sectors (i.e. NGOs, public authorities, river trusts, forest owners’ associations, companies, etc.) and can operate at local up to national and international scales.

Effectively linking users and providers through a PES scheme is challenging given the complex environmental, geographic, political, economic and social contexts in which PES operates. This reality motivates the potentially important roles of intermediary actors in PES design and successful operation (Moss et al. 2009). Intermediaries can possess the ability to work across the often impermeable boundaries between different actor groups, arenas of actions, or geographical scales which
have characterized the governance of these infrastructure systems in the past (Moss 2009).

As regards buyer(s) and supplier(s) there might be different combinations of these actors. We distinguish four types of market situations (Lockie 2013):

- **One to one** with only one/few ES sellers and one/few ES buyers.
- **One to many** represents a monopsony or oligopsony with many ES sellers but only one or few ES buyers.
- **Many to one** represents a monopoly or oligopoly situation with only one/few ES sellers but many ES buyers.
- **Many to many** represents a PES situation with many ES sellers and many ES buyers (polypoly). The combination of different actors and market types originates several PES governance models that are broadly characterised in the next section.
4. **Classification of Payment Mechanisms**

The type of actors influences the design and implementation of PES schemes. Following the IAD (Institutional Analysis and Development) approach (Ostrom 2011), a comprehensive analysis has to take into account different aspects such as: (i) preference and resource roles, rights and responsibilities; (ii) preferences, interests, expectations and values; (iii) actions and interactions, use and management of resources; (iv) information sharing; (v) lobbying; (vi) deliberation. These aspects are all useful to better understand the decision-making processes upon resource strategy and management. Actors’ interactions create networks and rules on resource management and conservation. Main PES classifications are based on the type of actors such as public, private, private non-commercial. In this regard, there are three broad types of PES schemes (Greiber 2011; Smith et al. 2013):

- **Public payment mechanisms** through which government pays land or resource managers to enhance ecosystem condition and services on behalf of the wider public. These are based on fiscal instruments (such as taxes or subsidies), relies on user fees, a government-driven system is established in which the public entity can play either as a provider or as an intermediary. In these schemes, the buyer is a third party (often in hierarchy) acting on behalf of service users, which “acquire(s) funding to compensate service providers through allocating revenues derived from earmarked tax revenues or general budget” (Porras et al. 2011). These kind of schemes are of a Pigouvian nature (Schomers and Matzdorf 2013). The participation of the end-users may not always be voluntary, as when all citizens are taxed regardless of their individual use of the service(s) provided. Such schemes are generally large in scope, provide legitimacy, and offer scale economies in transactions. On the other hand, government-financed schemes cannot always observe directly whether ES are provided, they do not have a direct incentive to ensure that the scheme is working efficiently, and they are likely to be subject to side-objectives such as meeting political pressures or alleviating poverty.

- **Private payment mechanisms** are self-organised private deals in which beneficiaries of ecosystem services contract directly with service providers. These schemes represent direct payments by service beneficiaries to service providers, in which both providers and beneficiaries are private entities (individuals, groups of individuals, private companies); the government can participate only as an intermediary. In these schemes, the buyers are the end-users of the services. These schemes can be seen as “private deals” and reflect consumer service demand. They usually operate at a small scale, and target only one or a few services (Porras et al. 2011).

- **Public-private payment mechanisms** that draw on both government and
private funds to pay land or other resource managers for the delivery of ecosystem services. These “acquire funding to compensate providers through allocating revenues derived from user fees or tariffs from a public utility or a regulated private utility” (Porras et al. 2011). Public-private schemes, a specific subset of private schemes, in principle have the same features as a private scheme, except that the buyer (or one of the principal buyers) is a public body. The feature which distinguishes public-private schemes from local public schemes is the role of the participating public utilities in public-private schemes. This role is limited to that of providing funds to the PES schemes in the role of a service buyer, just as any other private buyer would do. This means that the utility is not involved in the administration and management of the PES contract, as in local-public schemes, but participates as a contracting party of service buyers. In public-private schemes, the PES contract is thus administered by a third-party PES-management entity in the same manner as in private schemes.
PES schemes can be developed at a range of spatial scales, including international, national and local level (Smith et al. 2013). Public schemes may operate at the local or national level, and private self-organized schemes are typically local schemes.

Public and private PES schemes may adopt different financial arrangements regarding the compensation to sellers and the collection of buyers’ contributions. The most common financial arrangements include (for sellers) direct compensation, investment or development funds, and (for buyers) customer-charged payments, lump-sum contributions and tax-based contributions.
5. **Other Market-Based Mechanisms Used in Forestry**

There are several types of market-based instruments that can be differentiated by the degree of government intervention, the nature of the transaction and the characteristics of the buyers and sellers.

Support measures of various forms such as transfers, payments, assistance or protections are generally referred to as **subsidies** (eftec and IEEP et. al 2010). Subsidies or subventions are government payments to individuals or legal entities without being directly conditional on any defined output. The payments are connected to certain requirements and aim to reward desirable behaviour. Subsidies are paid to all subjects who fulfil the set requirements. The forms of subsidies are manifold, including direct payments, low interest rate credits, state guarantees or tax exemptions (indirect subsidies) (European Commission 2008). They may be justified to correct market failure e.g. to encourage socially beneficial behaviour. Subsidies can be environmentally friendly or environmentally harmful depending on what activity they are designed to support and the environmental impacts of that activity (eftec and IEEP et.al 2010). In forestry, they are used to support forest holdings economically (e.g. subsidising the construction of forest roads or other investments for rationalisation of forest production or innovation) or to guarantee the provision of forest ecosystem services. In Europe, subsidies are often given for measures of “multi-functional” or “close-to-nature” forest management. Specific purposes also include the protection of biodiversity, soil, air, water and recreational uses of the forest, as well as climate regulation, protection against natural hazards, landscape amenities, and historical and cultural sites. Subsidies may be granted by local, provincial, national governments or supranational/international levels (European Commission 2008).

**Public duties or taxes** are financial charges imposed on individuals or legal entities by the state. Besides of the financing function for the state, taxes also have the function to control certain behaviours by putting a price on resource use. A special case is the use of environmental taxes (eco-taxes, ecological taxation) which promote ecologically sustainable activities. Eco-taxes aim to correct negative market externalities by discouraging people from overusing resources. There are two main aims for environmental taxes. The primary goal is to use taxes to increase the price of products which are considered to be undesired products, in favour of more environmentally friendly alternatives that become more competitive in comparison, as a result of the tax increase on other products. The second goal is to finance the costs of collection and treatment systems or other compensation measures. This is a relevant measure in forestry because the collected funds may be invested back into forests in order to manage them for multiple / social benefits; thus, such charges, known as earmarked charges would finance specific purposes. Special forms represent tax exemptions. They differentiate
according to environmental impacts and honour environmentally friendly behaviour. Exemptions from tax duties can also be regarded a subsidy as some entities or behaviours are favoured. They are often used in forestry to compensate for legal restrictions on the use of forests, e.g. in protected areas (eftec and IEEP et. al 2010; European Commission 2008).

**Credit programmes** may be implemented as "bubble" scheme (a number of stationary emission sources are assigned a certain limit together), an "offset" scheme (firms buy pollution allowances from other firms that abate their emissions), or a "banking" scheme (where firms may store earned emission credits for future uses) (European Commission 2008).

Under a **cap-and-trade scheme** a cap is established for the use of a certain resource or the release of certain pollutants. The cap is the aggregate maximum amount of subtracted material or of pollution that can be released by participating entities. Tradable permits or credits are then allocated by dividing up the allowable overall total among those who participate in the established market. Industries or companies can sell permits that they do not need to other participants who need more than their allocation. This rewards companies which cut their pollutant discharge while it penalises those who pollute more heavily, and thus, creating an incentive for them to invest in pollution control. Trading increases the economic efficiency of resource management, by enabling companies or landholders to buy permits from those able to comply in a cheaper way (Smith et al. 2006; SFC 2008).

A typical forest-relevant application is the trade in greenhouse gases/carbon or emissions permit. Emissions may be reduced by cleaner technologies or other abatement measures. In the case of carbon emissions compensation is possible through carbon sinks or carbon sequestration. Forests act as carbon sinks as the carbon is stored in the woody biomass and other elements of the ecosystem (as long as they remain intact and are not destroyed by fire, or decay releasing the carbon stored back into the atmosphere, e.g. due to pests and diseases). The Kyoto Protocol has paved the way for governments and private companies to earn carbon credits which can be traded on a marketplace (carbon offsets) (European Commission 2008).

Markets can also be created for other forest-related services such as water and mineral extraction or biodiversity. An example that works on the basis of legal obligations for compensation of adverse impacts of development projects are **conservation banks**, in which projects with negative effects on the landscape or biodiversity pay banks or credits associated with new projects created by the banks in order to compensate for their unmitigated adverse impacts. The bank holds/purchases land on which projects are realised to balance the adverse effects to the environment. Sites are chosen and managed for their natural resource values and special-status species or sensitive habitats. Sites may be natural (preservation) and/or include restoration, and/or creation of habitat (White 2008; Carroll et al. 2008).

**Biodiversity offsets** are mechanisms to compensate unavoidable impacts of a project or plan on biodiversity through conservation or restoration actions. Most biodiversity offsets compensate for one or just a few dimensions of biodiversity, like species composition, habitat structure, ecosystem function or cultural values (Gonçalves et al 2015).

The simplest pure market mechanism for the financing of forest benefits is the **direct acquisition of goods** (such as timber, fuel wood, forest fruits, mushrooms, greenery, etc.) or **services** (such as catering, accommodation, education services or adventure, the right of access to the land or sport facilities, hunting and fishing rights, etc.). The list of goods or services that can be traded on markets is long; however, the trade is not always well developed and the goods and services are partly defined as public goods by law or according to their nature. Purchase of goods and services is an important category that depends strongly on the innovation and marketing activities of land owners (Mantau et al. 2001; Rametsteiner et al. 2005). In the field of forest goods, marketing could be further developed with better organisation of the whole value chains. With regard to the marketing of services many possibilities exist but these opportunities are rarely utilised by land owners. Territorial marketing is an innovative approach of marketing bundles of goods and services of a particular region (European Commission 2008).
According to the European Commission (2008) the purchase of land is one of the often used private financial mechanism in European forestry. It is potentially the most cost-effective option in some cases, but it is also very simple. It is an appropriate organisational solution if the production requires a specific way of management of the land and if the management know-how is on the side of the user. The characteristic of this mechanism is that the property rights lie in the hand of one person or legal entity. The future provision of the goods and services from the land is not dependent on markets or the will of providers. In many cases a certain piece of land is required for the provision of the desired service (e.g. recreation forest or water reserve) and only one party is interested (e.g. nearby municipality); in this case there is a double monopoly and no competitive market exists. The parties have then to agree on the basis of negotiations which strongly depend on the priorities of both sides (wish to sell on the side of the land owner or political mandate to buy on the side of the interested user). Another possibility represents land lease. In a lease one person pays a rent for the right to possess a property that belongs to another person for a certain time period. The rent may be paid one off but is typically given in yearly payments. Lease of forests is not as common as lease of agricultural land. Examples from forestry refer to specific uses of the land such as, for instance, for recreational purposes, sports facilities, use of drinking water sources, nature conservation or burial sites. Similar to land purchase, leasing is particularly suited for complex services or when the know-how of the production/land management is on the side of the interested party. These mechanisms are of importance in cases when the users have a high interest in a specific type of land management, e.g. in recreational forests, water reserves, or nature reserves under total protection (European Commission 2008).
**Sponsorship** is a business relationship between a sponsor who provides financing, resources or services and a party which offers certain benefits in return. It is a contractual agreement with mutual benefit and the value of the provided financing can usually be deducted from the company’s tax dues. Usually the commercial advantage is to establish an association between the sponsor’s image, brands or products and the sponsorship investment. Typically, the sponsor receives the right for advertisements or may also receive benefits for their staff. Sponsors may successfully be found for the support of events or projects that offer a high public visibility (e.g. large audience of an event or visitors of a site). Further, the audience should be the target group of the company in order to receive the intended advertising effect. The sponsoring of cultural or ethical projects, such as in the case of eco-sponsoring, usually works in the way that the money is used for charity and for projects in the public interest, such as biodiversity conservation or the preservation of beautiful landscapes. Eco-sponsoring is usually done by enterprises in order to illustrate their care for the environment and for sustainable development. Typical projects are afforestation, maintenance of natural monuments, nature conservation or restoration projects, nature-related sports or cultural events, environmental education, etc. There should be a connection to public benefits and an appropriate audience (European Commission 2008).

**Donations** are gifts given voluntarily and without return consideration. They are typically given for charitable purposes. Donations take various forms, including cash or other funds, goods and services including voluntary work. They may be given by organisations or private persons, including commercial organisations. In contrast to sponsoring, the firms donate without expecting direct benefits in return. In the case of donations, the charity purpose would be in focus. Forest-related donations are often given for the preservation of rare or beautiful trees or sites, recreational facilities, environmental organisations, or projects in development cooperation. Similar to sponsoring, it is not easy for forest holdings to attract donations. Projects that gain high importance on local level or are connected with specific purposes are still promising. A possible strategy may be to find partnership with influential local institutions or a well-established organisation that is interested in the project. Examples for donations for forestry projects are often found on local level (e.g. sponsorship of extraordinary trees or stands by private persons, or funding of educational activities by local companies) or in cooperation with environmental NGOs that raise funds for the purchase or lease of areas of specific interest for nature conservation (e.g. purchase of natural forests that are at risk to be developed for commercial purposes). The most commonly used is the symbolic sponsorship of one tree, one hectare, one species, etc. (European Commission 2008).

An innovative financing tool for nature conservation and landscape preservation is also **auctioning**. Elements of a certain landscape are offered by the land-owners and their maintenance can be secured by organisations or citizens for a certain time (Wensing and van Santen 2008).

**Certification** is the process of indicating through labelling that a commodity complies with a set of regulations governing the production process. As a market tool it creates niches, increases product recognition and/or secures market access. It can also be used to achieve social or environmental efficiency by defining minimum performance requirements (Taylor 2005). It is defined as the confirmation of a certain property or quality of a person, organisation, product or process by an authority or an independent party. The basic idea of **ecolabels or sustainability labels** is that consumers support through their responsible choices environmentally friendly and sustainably produced products. It means that consumers buy certified sustainable supplies and pay a premium for promised ecosystem benefits. The most important certification schemes in forestry refer to timber from sustainable forest management. The certification is an instrument for integrated nature conservation because ecological and social standards are requested by the certification systems. The certification should provide market benefits to the producer/trader in terms of higher prices or increased market shares. In Europe, two main certification schemes are particularly relevant: the FSC (Forest Stewardship Council, with around 100 000 ha certified forests worldwide) and the PEFC (Pan-European Forest Certification, since 2003: Programme
for the Endorsement of Forest Certification Schemes (around 200 000 ha) standards. For other forest goods, especially food, the certification of organic production may be considered. Organic standards do also exist for food production in the agricultural sector. Certificates of origin guarantee that a product has been produced in a defined region. In regional certification, often also certain quality standards are included, e.g. the sustainable management of the regional resources or the use of particularly ecologically valuable tree species (European Commission 2008).

Examples of payment mechanisms described above are available at the FOREST EUROPE web-based portal on FES: https://foresteurope.org/themes/forest-ecosystem-services/interactive-map/.
Before development of any PES scheme, it is important to have a suitable institutional background. Every PES scheme should follow the key steps of its development (Figure 8). For successful implementation of PES schemes it is necessary to approach each payment for specific service individually and identify the framework which should be followed. In some cases, it might be a necessary precondition to develop market by creating legal conditions and rules for trading, as well as defining controlling and supporting organisations (Viszlai et al. 2016).

An important aspect of PES development and implementation is the clear deal identification i.e. to clearly define who is the ecosystem service provider (seller) and who is the consumer (buyer), as well as the details of the good and its provision. To identify service providers is not usually difficult. They are often forest owners (private or public bodies) managing forest stands and ensuring the multifunctionality of forests and the provision of ecosystem services. The service buyers may be private or public bodies, representing the demand side of the services. Besides that, deal details in the agreement between the two stakeholders have to be negotiated. After successful development of these steps it is possible to implement PES (Brand 2002).

A PES scheme can focus on more than one ecosystem service. Those services being sold are then described as having been “packaged”. Ecosystem services can be packaged in three distinct ways (Figure 9) (Smith et al. 2013):

- **Bundling**: a single buyer, or consortium of buyers, pays for the full package of ecosystem services that arise from the same parcel of land or body of water. For example, an agri-environment scheme funded by government on behalf of the wider public.

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**Figure 8 – Main stages in PES development (Brand 2002)**
Layering: multiple buyers pay separately for the ecosystem services that arise from the same parcel of land or body of water; layering is also sometimes referred to as “stacking”. For example, an area of peatland is restored and yields a range of saleable ecosystem service benefits. The carbon sequestration benefits are purchased by a business, the water quality benefits by a water utility, the flood risk management benefits by the government on behalf of downstream communities, and the biodiversity benefits by a wildlife charity on behalf of its membership.

Piggy-backing: in this case, not all of the ecosystem services generated from a single parcel of land or body of water are sold to buyers. Instead, a single service (or possibly several services), is sold as an umbrella service, whilst the benefits provided by other services accrue to users free of charge (i.e. the beneficiaries “free ride”). For example, a business pays an upstream land manager for riparian restoration work to reduce the downstream flood risk to its bankside facilities. These improvements simultaneously improve water quality, enhance recreational values and provide habitat for wildlife. However, no buyers are found for these additional services and the benefits they provide are received at no cost to end users.
How PES Works in Practice

Understanding how PES mechanisms work in theory and in practice, and knowing their limitations, is crucial for exploiting their full potential as a policy tool for solving complex environmental problems we are confronted with (Prokofieva 2016).

For a PES scheme to work it must represent a win for both buyers and sellers. PES may be positive from a buyer’s perspective if the payments are less than those associated with any alternative means of securing the desired service. PES schemes may be positive from a seller’s perspective if the level of payment received at least covers the value of any returns foregone as a result of implementing the agreed interventions (Smith et al. 2013) (Figure 10).

It is important that the financing structure of any PES should be sustainable and sufficient. This ensures that the incentive to provide the ecosystem service remains even in the face of continued competing land uses. In order for PFES schemes to be successful, it is important to achieve a win-win situation for both sellers and buyers. The payment offered to forest owners or forest managers must exceed their opportunity costs8 (Engel et al. 2008). On the other hand, for buyers, it must be worth the money they pay. According to Smith et al. (2013), the minimum PES payment would be generally expected to at least cover any (private) return forgone by the forest manager as a result of reduced timber production. The theoretical maximum payment would represent the cumulative value of additional ecosystem service benefits that would accrue to the buyers. Many of these benefits are still hard to quantify.

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8 Opportunity cost also known as alternative cost, is the value of a choice, relative to alternative. It could be defined as the loss of potential gain from other alternatives when one alternative is chosen.
7. **Preconditions for Implementations of PES**

A number of preconditions for the implementation of PES emerge through the literature and practical experience:

- **Economic preconditions**
  PES has relevance where hard conservation trade-offs between the private interests of potential service providers (e.g. upstream landowners) and external beneficiaries (e.g. downstream water users) prevail. Landowner actions imply environmental externalities, which users may be willing to pay for. However, no PES deal is possible if users’ maximum willingness to pay (WTP) falls short of providers’ minimum willingness to accept (WTA) compensation. Such a situation would typically reflect that the perceived values of the service are lower than the estimated cost provision landowners are facing for deviating from their first best land-use plan. “Perceived value” and “estimated costs” would refer to expected monetary cost and benefits, but also be moulded by nonmonetary values (e.g. cooperative and governance benefits), perceived PES risks (of non-payment and non-delivery of services, respectively) versus business-as-usual risks (e.g. from fluctuating commodity prices), or the impact of land-use regimes (e.g. variable tenure security, (il)legality of use) (Wunder 2013, 2018).

Both service users and providers will also have to consider in their equation informational requirements and transaction costs (Wunder 2013).

- **Cultural preconditions (users and providers motives for action)**
  Even if the basic economics are right, PES can only work if there is the right culture of give and take: service users can get their act together to pay, and service providers feel motivated by receiving payments to deliver more services. These are not conditions we can automatically take for granted. Any pre-existing intrinsic motivations for good stewardship are not crowded out by extrinsic PES incentives. In other words, payment on balance needs to motivate ES providers to sustainably deliver more ES. (Landell-Mills and Porras 2002; Porras et al. 2008, Wunder 2013, 2018).

- **Legislative and institutional framework**
  As nition of services, enabling of contracts and payments, and avoid counterproductive or unintended distributional effects. Rules and institutions must have mechanisms to enforce contracts based on reliable contract law with good governance, and credible enforcement (Smith et al. 2006). Institutions define the roles of different actors in PES schemes, and are also important to facilitate transaction and reduce transaction costs, coordinate with other policies and mechanisms, set up insurance or other mechanisms to manage risks and provide related business services should the need arise. Furthermore, institutions provide vital direction for the valuation, utilization and conservation of ecosystem services, helping to avoid conflict between the conservation and use of natural resources (Vatn 2010).
Effective governance is needed to support the establishment of PES schemes via legislation, for example laws to implement a new public payment scheme (UNECE/FAO 2014).

**Ownership and tenure rights**

Probably the most restrictive of all PES preconditions is tenure clarity and security among service providers (Wunder 2013). For PES to work, ownership and tenure rights have to be clearly defined and recognized. Tenure is a generic term referring to a variety of arrangements that allocate rights to, and often set conditions on, those who hold land. Tenure regulates access to and use of resources (UNECE/FAO 2014).

Ownership refers to a particular type of tenure in which strong rights are allocated to the landholder. Tenure arrangements may involve exclusive access (when only one person or group has access), or different types of access for different groups of people at different times. The ecosystem service provider must hold the rights to the service as a condition for PES because if property or use rights are unclear, the buyer of the service cannot define the conditions of payment. In situations where the land is open access with no clear private, public or communal owner, PES is not the solution (Ostrom 1990; Vatn 2010). Where resource access and ownership are disputed, “buyers” have little incentive to participate in a PES scheme as there is no guarantee that they will get what they are paying for.

PES schemes are applicable to all situations where ownership is clear, however they are generally easier to apply to private forests and so are currently used on a larger scale in countries with predominantly private ownership. User-rights will also need to be respected, such as the right enshrined in the laws and traditional practice for the public to use the non-wood products and services of the forest (UNECE/FAO 2014).

**Stakeholders and negotiations**

The identification and participation of a number of key stakeholders is also necessary when making a PES agreement in forestry. As noted in the TEEB report, wide participation in decisions relating to PES design and implementation can help ensure transparency and acceptance and avoid the covert privatization of common resources (TEEB 2010). Forest stakeholder analysis could help guide negotiations towards an agreement which is socially and politically acceptable as well as institutionally feasible (Smith et al. 2006).

As forest stakeholders may include the general public, or specific groups within it, capacity building and appropriate support are needed to ensure that these potentially weaker voiced stakeholders are able to participate in negotiations, as successful PES schemes need a strong commitment by all parties (TEEB 2010). Advocates or representatives will be needed to ensure that the changes to forest...
use are acceptable to them, and bad publicity repercussions thereby minimized (Smith et al. 2006).

The variety of forest PES stakeholders and their interactions are present in Figure 11.

Forest PES schemes may be influenced by distribution of bargaining power between stakeholders, especially the service providers and the beneficiaries. This can affect who is included in the scheme, the way the money is shared, the rate of payment and the conditions set for service provisions and access (TEEB 2010).

As a result of this, completing a forestry PES deal may take a long time. Throughout the negotiations, the aim should be to form an agreement that specifies the design and rules for operating a payment scheme, which is effective, efficient, enforceable, transparent, equitable and sustainable (Smith et al. 2006).

**Monitoring, enforcement and compliance**

An effective monitoring and enforcement is necessary to ensure the continued functioning of the PES scheme, the delivery of the intended service and its measurement. Payments need to be clearly linked to good ecosystem condition and service provision and should be withdrawn if managers of the forest resource abandon management practices associated with the service. Monitoring data of the services and ecosystem condition at the site can help improve the targeting of payments (TEEB 2010).

How compliance will be determined and monitored needs to be decided in advance. Self-monitoring and monitoring by service sellers and buyers using agreed procedures is also an option but, whatever the approach, it is crucial to clearly delineate responsibilities for providing compliance and agree on sanctions in the event of noncompliance (Smith et al. 2006). As many of forestry PES schemes involve vast areas of land, agreements on sampling and degree of non-compliance or delivery failure will also have to be agreed.

Control systems are an essential part of any PES scheme. Where these are already in place, for example to promote sustainable forest management or agro-environmental schemes, establishing a PES can be easy. However, this is rarely the case and the establishment of new control systems is often required, usually adapted from already existing institutions and structures (Vatn 2010) and shaped to the PES scheme (Corbera et al. 2009; Primmer and Karppinen 2010).
Today, non-market goods and services of forests are often provided as positive external effects of sustainable timber production and through regulatory provisions. It is generally assumed that the societal demand would often be higher than the actual supply (European Commission 2008).

Nevertheless, there is a wide range of mechanisms for financing provision of forest goods and services that are in use, conventional regulatory and financing instruments often not seen as effective and efficient in all cases (Cubbage et al. 2007). Some mechanisms could be modified to cover goods and services other than those originally targeted.

In the past few years, a clear trend towards market mechanisms already can be observed. Market-based instruments (MBI) have been increasingly recognised as important policy mechanisms for achieving environmental protection goals. MBIs can be defined as mechanisms that encourage behaviour (management oriented to provide a range of ecosystem services) through market signals (i.e. prices) rather than through explicit directives. MBIs are also mechanisms that create a market for ecosystem services in order to improve the efficiency in the way the service is used (Viszlai et al. 2016). Therefore, the creation of markets and especially local contractual systems should receive increased attention and support. Situations where the market can play a role in the provision of goods and services should thus be stimulated, while maintaining public payment schemes (SFC 2008).

Researchers and policymakers have tried to develop new ways of compensating landowners for forest services (European Commission 2008). PES play an important role in the MBI toolbox and are recognized as a key MBI for achieving environmental protection goals (Snowdon 2015). Since many such market mechanisms are often still in the project/starting phase, monitoring of their performance, exchange of results and awareness raising is required for broader implementation (SFC 2008).

PES should not be seen as an end in itself but as a policy tool with several advantages and opportunities:

- One of the foreseeable advantages of the successful implementation of PES schemes is to maintain a sustainable supply of non-market forest services. PES can serve as incentives for the providers of forest services for managing forest following a multifunctional approach and keeping constant or increasing the supply of services without any loss. Forests can be managed in sustainable way, conserving the biodiversity and developing the multifunctionality of forest stands. And in this point appears the strength of PES schemes – the buyer of ecosystem services supports the ecosystem services provider by payment, which might compensate a shortfall in timber production. PES might play a role considering the increasing societal demand of non-marketed forest services (UNECE/FAO 2014).

- The voluntary character of PES can be considered as a weakness in some cases and still, in some other cases as a strength.
PES instruments, because of their voluntary nature, offer a less prescriptive and coercive approach and therefore may be a more feasible instrument in practice in some situations (Dunn 2011), especially it seems to be most effective in private PES schemes. Voluntariness provides flexibility in decision making. The voluntary nature of PES gives the opportunity to negotiate deal details between stakeholders without any restrictions and limitations (within the boundaries of legislation) (Viszlai et al. 2016). It represents an opportunity to engage previously uninvolved actors (especially in the private sector) in conservation activities. Their behavioural changes are promoted with positive incentives rather than coercion, more likely leading to transformational change (UNDP 2017).

Gómez-Baggethun et al. (2010) point out that the focus on payment schemes has contributed to attract political support for conservation, but also to commodify a growing number of ecosystem services and to impose market logic in tackling environmental problems. The easily understood PES arrangements have already been shown to be useful tools in raising awareness about environmental issues with the general public (UNECE/FAO 2014).

PES brings opportunities for actions on political-institutional systems and enterprise development for innovation and enhancement of the marketability, as well as the development of direct sales of previously non-marketed non-wood forest goods and services.

Funding for environmental protection in most countries is done by complex systems of tax, subsidy, penalty and budget. Compared to other resource management approaches, PES schemes are often recommended as being more flexible, more easily applied and more cost-effective, allowing high customization to local circumstances (Ferraro and Simpson 2002). PES makes a simple link between the use of an environmental service and the payment, which goes directly to providing it. Any system like this which can be easily grasped by the public, the media and opinion formers can be immediately seen to be ‘doing good’ in environmental matters – forests are saved (UNECE/FAO 2014).

Although PES programs are not designed for wealth redistribution, there can be important synergies with social aims when program design is well thought out and local conditions are favourable. This might specifically support the European policies for rural areas. According to Pagiola et al. (2005), PES offer distributional benefits, if communities can improve their livelihoods by offering and selling their ES and through access to new markets (Wunder 2005).

Besides providing funding to land users, PES schemes may also provide non-monetary benefits such as training, specialist advisors, infrastructure improvements or technical support. Furthermore, PES schemes bridge the interests of landowners, resource users and nature protection, and can therefore be seen as an efficient tool to address a set of problems. Rural communities can benefit from increased knowledge of sustainable resource use practices that are usually connected to PES through the provision of training and technical assistance. However, it is not well understood whether or not these potential benefits are realised in practice, or how they depend on scheme design (Thomson et al. 2014).

PES provides a potential platform to integrate conservation and climate efforts into a common policy framework, and facilitates the transition from an economy of production to an economy of stewardship (UNECE/FAO 2014).

Being involved in a PES offers a publicity boost for the companies involved. This is a benefit for the company involved, although it may mean that the reputation of PES may rise or fall with the reputation of these high-profile companies (UNECE/FAO 2014).

On the other hand, various difficulties and challenges can be recognized in the implementation of these new financing mechanisms, that can partly be explained by being in early stages of the innovation
process and by the weak support provided by the institutional system. The coverage of the initial costs of such initiatives also often forms a bottleneck (SFC 2008). However, we should not forget that we still operate in a field where the marketability is and will stay restricted, at least to a certain extent. The most common risks are associated with following:

> The definition, understanding, measuring and economic assessment of ecosystem services at appropriate scale and precision remains a basic challenge for the implementation of payment mechanisms. While this requires appropriate scientific knowledge and technical competences and skills, it also builds on stakeholder consultation. Besides the site-specificity of services, the sharing of knowledge and experiences can help reducing costs and promoting a more efficient approach to the study of ecosystem services and the implementation of payment mechanisms. Information can also allow the development of an accounting system focusing not just on ecosystem service flows, but also on the natural capital (stock) (Leonardi et al. 2016).

> A pre-requisite for establishing a payment scheme is the existence of institutional and political support. The application of a specific payment scheme depends on the interest and willingness of involved actors, laws and regulations in place and sufficient financial resources. In general, society may be willing to pay for non-wood forest goods and services, but operational mechanisms supporting valuation and financing remain comparatively rare due to low interest or limited information and are not fully reflected in forest policy. These shortcomings should be made more explicit to policy-makers while exchange between countries/regions on practical examples could be used to devise clear guidelines for implementation of successful mechanisms (SFC 2008).

> Constraints in the creation of markets are often related to high transaction costs (include the cost of identifying and selecting service providers, attracting potential demand/buyers, negotiating and developing contracts, training, monitoring, reporting and follow-up activities, etc.) and/or the legal and socio-economic framework, such as the open access to forests and everyman’s right. Also, people may misperceive the distribution of ownership rights of non-wood forest goods and services and expect to use them for free even when this is not legally the case, or they may regard them as valuable but expect them to be paid from public budgets (SFC 2008).

> Another potential problem for the implementation of PES is weak ownership and tenure rights of forest land. Forest tenure must be clearly defined and recognized and the ecosystem service provider must hold the rights of the service as a pre-condition for PES. This is because if property or use rights are unclear, the buyer of the service cannot define the conditions of payment. This condition is strongly connected with forest and other wooded land ownership (Viszlai et al. 2016). Changes in land management rules and regulations may also have a significant impact on ecosystem service delivery and the PES.

> Failure to monitor the effectiveness of the compensation schemes, including risks of not fulfilling the performance condition. Inappropriate or absent monitoring and evaluation of PES is commonly referred to as a strong limitation to identifying both their direct and indirect impacts on both human (i.e. socio-economic) and environmental systems (Asbjornsen et al. 2015). Yet, unless contract compliance is both credibly monitored and enforced, contracted landowners may receive payments while continuing business as usual, that is, profitably defecting on their contractual obligations. Monitored and enforced conditionality is necessary to make PES function as effective incentives for conservation (Wunder et al. 2018).

> There is also the concern that tapping new income sources is particularly difficult for small land owners who may lack the resources for developing the necessary marketing skills, cover their administrative burdens, etc. This calls for special attention and possibly additional support from governments, land owners and NGOs alike when attempting to apply MBI.
It is also argued that PES may become counterproductive. Assume that the service was supplied as a matter of course and as a social obligation for free. When a system of payment is introduced to guarantee quantity and quality of service, the logic has changed. If the payments are now seen as insufficient, appeals to social obligation will be useless.

A number of successful examples for the application of PES mechanisms seem relevant, promising and feasible for the support of forest goods and services but their potentials are still not fully utilised and studied. Their real potentials and limitations can therefore not be assessed reliably. The lack of knowledge includes questions regarding the role of institutional actors in the development of MBI and in the support of innovation processes. It seems that improvements not so much depend on the development of new mechanisms but more on an increased use of knowledge and established mechanisms and their proper implementation (European Commission 2008).

Whilst the emphasis of PES has always been on improving the quality and sustainability of environmental systems, it would be easy to label the contributions of companies as conscience money, paying for irreplaceable environmental damage. It is also sometimes argued that PES schemes can be unfair and can provide perverse incentives where payments go to those who have degraded or threaten to degrade their land, rather than those already sustainably managing it. It will be the job of any future PES scheme to address and allay such fears which will undoubtedly arise. Trading schemes will be particularly vulnerable to this criticism (UNECE/FAO 2014).

The ecosystem service paid in the PES scheme may not be the most vulnerable, or most vital, service in the region, however it will benefit due to its fortunate proximity to an identifiable user. PES tends to favour environments located in populated regions rather than, remote areas which may be under more environmental stress (UNECE/FAO 2014).
PES policies represent a growing trend in conservation policy. By altering private incentives to induce desired outcomes, PES schemes offer a direct, and possibly more equitable, method for achieving environmental outcomes than other approaches. However, the context in which a PES initiative is implemented matters greatly for effective policy design and the achievement of stated goals.

Whilst the above has made the case for the usefulness and application of PES, it must be acknowledged that this approach does not exist in a vacuum and will need to ‘win the hearts and minds’ of the governments, private sector and the general public in the countries where it could be adopted (UNECE/FAO 2014). The importance of context in achieving policy goals emphasizes that no single policy exists which would suit every scenario. Previous experience with incentive-based approaches suggests it is unlikely a PES approach will always be able to simultaneously improve livelihoods, increase ecosystem services, and reduce costs. Potential trade-offs among these goals can be assessed reasonably well by considering the correlation between characteristics of poor landholders and their land, characteristics of the costs and benefits of providing ecosystem services, and the political feasibility of various policy options (Kosoy et al. 2007). Special attention should be paid to securing tenure rights, because land-use is often the basis for schemes which normally compensate a restriction of land-use or finance specific management measures on a specific type of land (UNECE/FAO 2014).

Current knowledge and experience also suggest other areas in which additional research is needed. Several PES projects that have been running in developing countries for some time are starting to offer promising findings about the use of PES mechanisms (Rosa et al. 2003; Sanchez-Azofeifa et al. 2007). However, new projects will only be able to learn from the successes and failures of their predecessors if the manner in which outcomes relate to the environmental, socioeconomic, and political contexts of the policy are systematically documented and compared across a range of cases. More long-run experience, rigorous program evaluation will provide additional understanding of the effectiveness of different policy designs over time (Ferraro and Pattanayak 2006), as well as information on how PES schemes respond to exogenous shocks. Collaborations between ecologists and economists can better specify the production function for ecosystem services. Communication actions should not only be directed at ecosystem providers or buyers; they also should target decision-makers and the general public because political support is often needed, especially during the early phases of development. Pilot projects are often a good way to demonstrate the relevance of PES and show results (UNECE/FAO 2014). This information will improve the design of input proxies and reduce the uncertainty surrounding environmental effectiveness. More research is also needed on how incentive-based mechanisms can account for potential trade-offs and synergies in the production of multiple ecosystem services. Additional analysis of large-scale PES policies can help us to understand the broader effects on the economy from scaling-up PES schemes (Xu et al. 2006; Sullivan et al. 2004).
Despite the fact that more than 50% of the forests in Europe are privately owned, depending on national legal frameworks, the access to and the use of the majority of non-wood forest goods and services is free for the public. So, while the benefits of environmental services are public goods, the cost of ensuring their provision often falls on local land managers (Gutman 2006). Some people, however, regard them as valuable but expect them to be paid from public budgets. This can cause market failure and has considerable implications for valuation and values as well as payment mechanisms and financing. Payment schemes may be required where scarcity and perhaps decline in availability are evident and where existing incomes to forest owners are insufficient to maintain or maximise the provision of goods and services (SFC 2008). The expansion of pricing mechanisms to ecosystem services has followed two main approaches.

The polluter pays principle is grounded on an alleged ethic of responsibility, according to which the economic agents causing environmental harm should carry the economic costs of the negative externalities they create. Since the 1980s the polluter pays principle has been incorporated in legal texts. In Europe it was included in the Single European Act of 1986 (article 174), in the Maastricht Treaty (article 130.2), and in the Treaty establishing a Constitution for Europe (article III, 233.2), which has stagnated since 2004. At international level, the polluter pays principle was adopted by the Organization for Economic Cooperation and Development (OECD) in 1972, and was contemplated in the Declaration of the Rio Summit on Sustainable Development of 1992 (article 16). Since the 1990s the leading instrument used to operationalize the beneficiary pays principle are the so-called Markets for Ecosystem Services (MES).

If negative environmental externalities are addressed through the polluter pays principle, positive externalities are dealt with through the steward earns principle, as an underlying logic for making payments for ecosystem services. The underlying rationale is that beneficiaries of ecosystem services should compensate the stewards that maintain or protect the services from which they benefit (Gómez-Baggethun and Ruiz-Pérez 2011). The widespread expansion of PES as integrated development and conservation scheme, however, dates fundamentally from the last two decades (Wunder et al. 2008).

The concept of commodification refers to the expansion of market trade to previously non-marketed areas. It involves the conceptual and operational treatment of goods and services as objects meant for trading. It describes a modification of relationships, formerly unaffected by commerce, into commercial relationships. Commodification of ecosystem services thus refers to the inclusion of new ecosystem functions into pricing systems and market relations (Gómez-Baggethun and Ruiz-Pérez 2011). It takes place though four main stages: economic framing, monetization, appropriation and commercialization.

The first stage consists of the discursive economic framing of ecosystem functions as ecosystem services, which started with the anthropocentric interpretation of ecosystem
functions and continued with the application of the ecosystem service concept from the 1960s (Gómez-Baggethun et al. 2010).

The second stage takes place when the use values embedded in ecosystem services are expressed as exchange values through monetization or pricing. The conceptual roots of this process in economic theory have been traced back to the early 19th century (Gómez-Baggethun et al. 2010), but relate more directly to the origins of the externality concept coined in the 1920s (Pigou 1920) and even before (e.g. Dupuit 1952, quoted in Maneschi 1996). Although economists have tried to attach monetary values to ecosystems since the late 1950s (e.g. Clawson 1959; Krutilla 1967), environmental scientists did not pay much attention to this work until the 1990s, when they systematized valuation frameworks (Bateman and Turner 1993; Freeman et al. 2014; Heal et al. 2005; Pearce 1993; Pearce and Turner 1990; Turner et al. 2004). Finally, after the publication of the much-discussed paper by Costanza et al. (1997) that estimated the total worth of the Earth’s natural capital, valuation became one of the most frequent target of ecosystem services research.

The third stage consists of the appropriation of ecosystem services, and operates through the formalization of property rights on specific ecosystem services, or on the lands producing such services. This stage has often involved privatization, through which ecosystems that were previously in openly accessible regimes, or in communal or public property regimes, have been turned into private property. Although the origins of this process can be traced back several centuries (Ingold 1986) the direct theoretical roots of the recent cycle of the privatization of nature lie in the influential contributions of Coase (1960) and Hardin (1968). The defence of the former for well-defined property rights was complemented with the advocacy by the latter for privatization (or alternatively appropriation by the state) of common pool resources as the way to avoid overexploitation.

The last stage in the commodification process consists of the commercialization of ecosystem services - i.e. the creation of institutional structures for ecosystem services sale and exchange. As with any other monetary market, MES and PES involve the definition of one or more services, which then become commodities subject to trade. The extension of MES and PES towards new ecosystem functions therefore involves, by definition, a process of nature commodification - i.e. an expansion of the commodity frontier into previously non-marketed spheres of the environment (Kosoy and Corbera 2010). The commodification process is finally completed with the implementation of institutional structures allowing for transactions in market exchanges, as occurred with the establishment of MES and PES schemes.

Payment for ecosystem services (PES) is an example of an innovative and relatively young market-based mechanism (Smith et al. 2013). PES, originally meant as cash transfers, include all financial and non-financial rewards (or compensation mechanisms) between buyers and sellers for the provision of ecosystem services (Wunder 2007). Clearly, PES is a market tool through which the public sector can directly and actively enter the green market and become a “buyer” of ecosystem services. The basic principle is that those who “provide” ecosystem services should be rewarded for doing so (Gutman 2006).

There is a wide range of mechanisms for financing provision of forest ecosystem services that are in use. Most implemented payment schemes are public, followed by private mechanisms (market).

Situations where the market can play a role in the provision of goods and services should be stimulated, while maintaining public payment schemes, because state financial resources are often not sufficient to be effective in all cases. Creation of markets and especially local contractual systems should receive increased attention and support. Many such market mechanisms are often still in the pilot project/starting phase. Since they are likely to remain important in the future, monitoring of their performance, exchange of practical examples and results between countries/regions, as well as awareness raising is required for broader implementation of successful mechanisms (SFC 2008).

The application of a specific payment scheme depends on the interest and willingness of involved actors, laws and regulations in place and sufficient financial resources. Population pressure and access to a limited forest resource...
may drive interest from the user to secure through payments access to forest ecosystem services, while lower timber prices and reducing incomes available to maintain the resource in a good state may raise interest from the side of the forest owner. However, in general, society may be willing to pay for non-wood forest goods and services, but operational mechanisms supporting valuation and financing remain comparatively rare due to low interest or limited information and are not fully reflected in forest policy. These shortcomings should be made more explicit to policy-makers, because a pre-requisite for establishing a payment scheme is the existence of institutional and political support (SFC 2008).

Before PES policies and programmes are introduced, policy planners need to investigate and address potential adverse impacts and the context specific question of external agents' legitimacy to change property regimes, social relationships and natural resource values in the targeted areas. It is important to take account of existing historical contexts, local institutions, the distribution of rights to natural resources, and internal and external pressures on sustainable practices of rural communities, i.e. pressures emanating from the political economy imposed on developing countries, agricultural prices, land pressures, development practices, etc (de Francisco and Boelens, 2014). Indeed, their success depends in great part on pre-existing conditions (Gutman 2006). There is no single, transferable model for PES systems. They are highly adaptable and several different models currently coexist in different markets and locations. Each one must be tailored to the specific conditions of the market for a given environmental service in a given location (Mayrand and Paquin 2004).

PES systems work best when services are visible or quantifiable and beneficiaries are well organized, and when land user communities are well structured, have clear and secure property rights, and strong legal frameworks. These conditions minimize sources of interference with the newly created market and reduce transaction costs. This suggests that part of the success of schemes rests in the selection of regions/communities where they will be implemented or on work conducted in their preparatory phase (Mayrand and Paquin 2004).

In conclusion, PES schemes have the potential to become very valuable transfer mechanisms for internalizing positive environmental externalities and generating new revenues for sustainable development, as well as encouraging and financing conservation efforts. This is especially important in the current era of increased funding needs for the environment, as well as in the securing of global commons. PES may also succeed where other conservation approaches have failed (Gutman 2006). This potential will be gradually fulfilled as markets for environmental services mature over time and as PES schemes become more financially sustainable (Mayrand and Paquin 2004). To achieve a better recognition of forest ecosystem services and extend financing and marketing efforts, land owners and managers, their interest groups and extension services need to raise more strongly the awareness of decision makers and the general public on this issue (SFC 2008).
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PART III.
FOREST EUROPE
Web-Based Portal on Forest Ecosystem Services
Background

At the 7th FOREST EUROPE Ministerial Conference held in October 2015, in Madrid, the European ministers responsible for forests signed the Madrid Ministerial Resolution 1 “Forest sector in the center of a green economy”, and expressed their commitment inter alia to “Promote the exchange of information on methodologies and practices on the valuation of and payments for forest ecosystem services as well as policy approaches to this end”.

This Ministerial commitment was subsequently reflected in the FOREST EUROPE Work Programme 2016–2020, action 4.4.2 “Promotion of the Pan-European practices on valuation of and payments for forest ecosystem services”. In line with this action “a web-based portal shall be established to serve as a platform for knowledge and information exchange on valuation methodologies of and payments for FES as well as sharing best practices in this field”.

The FOREST EUROPE Web-based Portal on Forest Ecosystem Services is a tool that aggregates available and published information on valuation of and payments for FES in one place via simple browsing systems. It helps to facilitate decision-making on FES and select an appropriate method for their valuation as well as appropriate financing mechanisms. Policy makers will also have the opportunity to explore existing valuation methods and payment schemes implemented in FOREST EUROPE signatory countries. The web-portal also provides a set of recommendations, outlines, opportunities and risks associated with valuation of and payments for FES.

The portal comprises three modules: (i) Introduction to forest ecosystem services; (ii) Valuation of forest ecosystem services; and (iii) Payments for forest ecosystem services.

The FOREST EUROPE Web-based Portal on Forest Ecosystem Services is accessible at https://foresteurope.org/valuation-forest-ecosystem-services/.

Figure 12 – Home page of the Web-based portal on FES.
Description of Module I

Introduction to Ecosystem Services

The first module contains a brief introduction to ecosystem services providing a conceptual framework for ecosystem services and clarification of three interlinked concepts related to the provision of ecosystem services, i.e. ecosystem process, ecosystem function, and ecosystem service. This module also refers to comparison of three main approaches to classification of ecosystem services according to (i) the Millennium Assessment (2003), (ii) TEEB (2010), and (iii) CICES V 4.3. elaborated by the former FOREST EUROPE Expert Group that worked in 2012–2014.

Description of Module II

Valuation of Forest Ecosystem Services

Despite the fact that a number of studies have been conducted to estimate a value of the forest ecosystem services which have applied a range of various valuation approaches and methods, many of them are not publicly available or difficult to understand because of different languages or sophisticated terms.

The second module adopts interactive schemes developed by the Expert Group on FES which allow simple orientation within different valuation methods and access to their case examples collected from FOREST EUROPE signatory countries.

Case examples published at the web-portal were elaborated by compiling summaries of valuation studies following a pre specified template describing (i) scope of the study—description of valued ecosystem services and geographical scope covered, (ii) valuation method(s) applied, and (iii) key results of valuation study.

Case examples were elaborated by invited experts from the Austrian Research Centre for Forests (BFW), FOREST EUROPE Expert Group members and Liaison Unit Bratislava using various data sources such as background information from these countries.

Figure 13 – Module I - Introduction to Ecosystem services with case examples of implemented FES.
documents provided by FOREST EUROPE signatories and observers, valuation studies available in Web of Science or Scopus databases, a review of literature sources, and outcomes of national or international research projects. Browsing within this module is possible in two ways. Firstly, it is possible to search for valuation approaches and methods according to type of FES to be valued. This helps relatively quickly and clearly determines which method/s should be used to value individual FES (Figure 14).
Secondly, it is possible to obtain an overall overview of approaches and methods used for economic valuation (Figure 15). Both interactive schemes also show simple references to the explanatory notes where description of the method, its benefits and limitations as well as case examples are shown after clicking on the method title.

Case examples are provided especially for the contingent valuation method, choice modelling, the value transfer method, the market price method, the travel cost method, and the hedonic pricing method. Other methods listed in the interactive scheme contain only descriptions based on the literature review without case examples. This is because the respective method is only rarely used (most probably because it is not practically applicable), or because there is an overlap with another method (some valuation methods have several variants, and/or sub-categories, which are in fact almost identical, and are being used for essentially the same purpose).

Description of Module III
Payment for Forest Ecosystem Services

The third module on payments for forest ecosystem services includes a short introduction on the topic and a description of basic payment mechanisms. To illustrate the implementation of PFES in the pan-European region and to promote exchange of information and practices on payments for FES, an interactive map of FOREST EUROPE signatory countries has been developed since 2018 (Figure 16).

Users of the interactive map are able to select case examples of PFES by using the following criteria: (i) country, (ii) financing mechanism, and (iii) type of forest ecosystem service.

To collect case studies, representatives of FOREST EUROPE signatories were approached with a request to contribute to mapping of PFES implemented in the pan-European region and built up the interactive map. Each case example
follows the common template describing (i) location where PES was implemented, (ii) compensated ecosystem services, (iii) PFES functioning, (iv) actors involved in PFES, (v) time horizon, and (vi) availability of economic data. There are various examples of well-functioning and/or developing PES schemes in the forest sector in FOREST EUROPE signatory countries. These examples represent a basis for development of new PES. However, identification and assessment of local socio-ecological conditions needs to be taken into account.
Analysis of Different Approaches and Methodologies on Valuation and Payments for Forest Ecosystem Services in the Pan-European Region

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