



Novel business models and mechanisms for the sustainable supply of and payment for forest ecosystem services

WP 3 – Methods and models to assess forest ecosystem services

Task 3.3: Methods to assess economic value of ES

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Abstract

The demand for non-marketed forest ecosystem services, like for example, carbon sequestration has been increasing the last decades and this has been accompanied by attempts to establish new market for such services. This development is an opportunity for forest owners to increase their rentability of their forest management. It will also provide the beneficiaries of the services an instrument to influence the forest owners to increase their supply of services which, without new markets, would be considered as externalities and therefore often ignored in their management. This concerns both governmental funded payment schemes, crowd-funding projects, or companies use of forest project in their corporative social responsibility activities. To increase the transparency of transactions between forest owners and buyers of services and to better assess the potential of such new markets, it is important to estimate the demand for services. Likewise, it is also important to assess the cost of increasing the supply of services to increase the transparency and also to assess additionality of projects. In this report we present and discuss different methods which have typically been used in assessing the preferences for forest ecosystem services or to estimate the costs of provision. Besides a brief introduction to the methods the report also makes references to more detailed technical guidelines and key scientific references. Finally, the report also gives an example of how a change in forest management may influence the values of carbon sequestration and recreational use of a forest project in the French NOBEL pilot demonstrator.

1. Introduction

Forest ecosystem services as public goods

Forests provide a large number of different services which contributes to the welfare of the human society. From a production point of view a forest can be considered as a multiple-output production unit (Johnston, 2022; Lambini et al., 2018) where the objective of the forest manager is to choose a production plan which maximises the profit or utility given the prices of the different outputs. For some of the services provided by a forest there are markets (timber markets, hunting licences etc.) while for others there are not always well-functioning markets and therefore no prices (Garcia et al., 2018; Müller et al., 2020; Zanchi and Brady, 2019). Missing markets are typically due to the public good character of the services. For example, it is impossible to exclude people, who do not pay their share of the cost of increasing the carbon sequestration in a forest, from the benefit of climate change mitigation. While one could imagine that outdoor recreation in forests could be marketed, for example demanding entry fees, it would often be disproportionately costly to implement and enforce an entrance fee. Therefore, even though it may be technical possible it would not lead to a socially optimal use of the forest.

Incentives for provision of forest ecosystem services

It is well-known that there are no direct economic incentives for forest owners to consider the benefit non-marketed services represent for their users. Therefore, the forest owner may chose a management that enhance the provision of services that can be sold on the market on the cost of the provision of non-marketed services. This is the case when production of services are characterised by trade-offs (Borges et al., 2014; Eggers et al., 2019; Lambini et al., 2018). This is, for example, the case if intensification of the timber production (e.g. fast-growing monoculture forest stands) reduces the recreational quality of the forest. However, there may also be situations where provision of timber may be independent or complementary to other services and therefore no conflict between the supplies of different services. Whether we have trade-offs or complementarity in provision is an empirical issue which depends on the specific forest and situation.

In the project NOBEL a main objective is to suggest new market mechanisms that can facilitate the optimal supply of forest services. This objective relies on the hypothesis that for the supply of one or more forest ecosystem services is suboptimal due to lacking markets. By establishing new markets, services which hitherto were not considered by the forest manager will get a market price. This will give the forest owners incentives to account for these services in the management of their forests. If such new markets are well-functioning, it is a general result in economic theory that the supply will be socially optimal. Such an optimum will normally imply that the marginal cost of provision equals the marginal demand of the services. However, such new markets are only likely, in the best cases, to be approximations of a well-functional market. Dysfunction of markets are due to, for example, informational deficits, transaction costs, uncertainty, owner structures, long time horizons and non-convexities in provision (e.g. economics of scale and complementarities in provision).

The use of economic costs and benefits estimates

This report gives an overview of the valuation methods which can be used to assess the cost of provision and the social demand for ecosystem services as well as addressing important aspects related to specific contexts of the NOBEL project (e.g. valuating new market mechanisms). Such assessments serve multiple purposes (Müller et al., 2019):

1. Assessments serve to evaluate the social optimal supply of a given service at a given location and time (Zanchi and Brady, 2019). Such value estimates will allow comparing the current provision with the socially optimal provision and consequently evaluate the need for policy intervention or the potential of establishing new market mechanisms.
2. Assessing the cost of provision may also provide important input to guidelines for policy makers considering implementing new policies. This includes the design of public-funded payment for ecosystem services programs, including setting subsidy levels.
3. An assessment of costs and benefits would also be a basis for assessing the distribution of benefits between sellers and buyers and the distribution within the two groups. While in perfectly functioning markets the marginal costs and benefits would be equal, in reality we may have examples where marginal benefits may exceed the costs of provision. In such cases the sharing of benefits between users and providers will often depend on the market mechanism implemented. This is not only important from a fairness or equality perspective but also potentially important from a “marketing” perspective. Private agents are less likely to pay a forest owner for a change in management if she or he is excessively compensated (Abildtrup et al., 2012). This would also contribute to the assessment of the additionality of given policy or action, i.e. ensuring that a payment changes the behaviour for the forest owner relative to the current optimal management, cf. (Engel et al., 2008): *“The (...) problem of paying for adoption of practices that would have been adopted anyway is known as a lack of additionality, or “money for nothing”.*
4. Knowledge about the costs of provision of ecosystem services represents important information for forest owners and managers. Participating in new markets would require management plans and actions which are new and therefore require new information from the forest owner perspective. This concerns both information of costs of production and the potential market value.
5. The spatial heterogeneity in the cost of provisions as well as benefits is important for the design of markets and policies. This includes the need for spatial targeting of policies or innovation activities. Therefore this report will also consider how assessment can be carried out in a spatially explicit manner, including benefit transfer (Johnston and Rosenberger, 2010) and ecosystem services mapping (Schägnler et al., 2013).

Objectives and scope of report

Objectives

The two main objectives of the present report are to provide:

1. an overview and an assessment of economic valuation methods relevant for estimating the cost and benefit of forest ecosystem provision
2. an overview and an assessment of methods and databases for carrying out benefit transfer of forest ecosystem services in an European and spatially explicit context.

Scope

There is a wide range of methods for assessing ecosystem services (Harrison et al., 2018; IPBES, 2015). However, this report is focusing on monetary assessment. Furthermore, we will not consider multi-criteria methods even though these are useful methods for analysing the supply of multiple services from a forest (Müller et al., 2019). Multi-criteria methods, including Pareto-frontier methods (Borges et al., 2014; Eggers et al., 2019; Marto et al., 2018), are powerful to analyse trade-offs between services and questionable economic valuations can be avoided. However, for identification of social efficient management scenarios it will require the definition of weights for the different criteria. Such weights are often subjective or based on expert assessment and may therefore not reflect the social demand (Zanchi and Brady, 2019). However, multi-multicriteria methods may also be useful to estimate shadow prices of (marginal loss in income due to constraint on non-market ES supply) for non-marketed services while not representing a preference-based (economic) value of the service in question.

Other guidelines

It is not the objective to provide an extensive guideline for the application of various relevant methods. The focus is on the relevance of different methods in the context of the NOBEL project, i.e. providing support for development of new market mechanism. For specific guidelines and for economic theory underlying these methods we make reference to relevant textbooks (e.g. Champ et al., 2017; Johnston et al., 2015) and methodical guidelines, e.g. for discrete choice experiments (Mariel et al., 2021). There are also a number of existing guidelines on valuation of ecosystem services available that can serve as inspiration. For example, (DEFRA, 2007) makes an introduction to valuation of ecosystem services in general while (Riera et al., 2012) provide a good practice manual for valuation of forest ecosystem services. An recent and comprehensive report is published by FAO which consider forest ecosystem services broadly and have a global perspective with examples from different continents, however, without addressing changes in forest management (Masiero et al., 2019). Binder et al. (2017) provide a non-technical review of the science related to assessment and valuation of forest ecosystem services. This report focuses on the US forests and includes both the quantification and valuation of services while Binder et al. (2018) review methods for valuation of ecosystem services provided by forests in UK. The European Forest Institut (EFI) published a report on valuation of the cost and benefit of positive forest externalities in the context of European forests based on the NEWFOREX project (Thorsen et al., 2014). An report published by (UNECE/FAO, 2018) focuses on forest and water and include, among others, an introduction to and examples of valuation of forest ecosystem services related to water. TEEB (2013, 2010) provides an extensive introduction to the principles and methods for ecosystem services valuation while not focusing directly on forest ecosystem services. A short introduction to different valuation methods can also be found for a great number of different methods on the IPBES site¹, including the forthcoming report on forest values where monetary valuation is only considered one out of many methods to obtain ecosystem values (IPBES, 2022). For a more general introduction of the economic analysis of ecosystem services see (Johnston, 2022).

¹ : aboutvalues.net/method_database/

2. The economic framework – key questions

In this section we will highlight some of the key questions to consider in the economic valuation of ecosystem services.

Marginal or non-marginal changes?

An important question to consider when carrying out economic valuation is whether we analyse marginal or non-marginal changes of service provision. By marginal changes we understand changes in provision which are relatively small relative to the current provision or consumption of services. In this case we will normally assume that relative prices are not changing, i.e. the price of, for example, one cubic meter of timber or the willingness-to-pay for one ton of carbon sequestration does not change by the changing forest management. However, this assumption would not be valid if the changes in management are substantial and all forest owners in large area, e.g. in Europe, change management. This may, for example, induce a substantial change in the supply of timber and thereby prices may change or may change the marginal costs of carbon emission reductions. In this report we will only consider marginal changes that will allow us to keep prices fixed. This is justified by the assumption that new markets, at least in an initial stage, will only marginally influence the supply of timber and other services². It is important, however, to have this assumption in mind and addressing non-marginal changes may be relevant in future work. This also implies that we are not addressing the “total value” of ecosystem services as it has been attempted to estimate in, for example, Costanza (2014, 1997) and Grammatikopoulou and Vačkářová (2021) for European forests. Such estimates have also criticized for not being meaningful (Howarth and Farber, 2002).

What is valued?

Physically quantification of services

In all economic valuations it is important to be explicit about what is valued? Often valuation studies have used proxies for the service provision or it is the ecosystem and not the services which are evaluated. The lack of clear definitions of the services evaluated reduces the value of results for policy makers (Keeler et al., 2012; Wong et al., 2015). How to quantify ecosystem services in quantitative terms and how to link values is a research topic in itself (Jensen et al., 2019; Johnston et al., 2016; Keeler et al., 2012; Wong et al., 2015).

Market versus non-marketed services

An important distinction is, evidently, if the services are marketed or not. While there may also be issues related to estimation of the prices for marketed services we will here focus on non-market services.

² In the case of recreational services, where users often are local residents, the changes in access to one site may influence the willingness-to-pay to have access of nearby substitute sites, which should be considered.

The value for who?

Methods for estimating value will depend on the perspective, i.e. whether it the user or the provider perspective. Are we estimating value for regional, national, and regional population? Results will often be very sensitive to such questions. It is also important to have in mind if it is the economic value that is estimated or the financial value. In estimation of the economic value, economic transfers (taxes, subsidies etc.) are ignored, implying taxes and subsidies should be excluded from the analysis while included in a financial analysis (Valatin et al., 2022).

Time perspective and discounting

Forest is characterised by long-term investments raising the question of which time horizon to choose for the analysis and the question of discounting, i.e. how should future cost and benefit be included in the assessment (Zanchi and Brady, 2019). An overall recommendation would be to use national recommended discount rates for project evaluation if the perspective is social cost-benefit analysis while long-term alternative cost of capital if a business perspective. These could also include risk premiums. It is in any case important to carry out sensitivity analysis with respect to the discount rate.

3. Economic valuation methods

Assessment of the cost of provision

This section draws on the work carried out in the NEWFOREX project (Thorsen et al. 2014a, 2014b)³.

The components of the cost of provision

The costs of the provision of ecosystem services have different components, including direct costs, indirect costs, transaction costs, so-called feedback costs. These different categories of costs will be discussed in this section.

Direct costs

Generally, direct costs, or implementation costs, correspond to the easiest measurable costs. They are linked to the undertaking of a specific management action for the provision of one (or several) ecosystem service(s): equipment for recreation, specific costs of management for maintaining (or restoring) biodiversity. The total costs incurred in the specific action for the provision of a service include operation costs and/or investments (costs of capital). The former are costs relative to the “maintenance” of the service provision. Actions that generate these costs have in general to be renewed each year. Instead, costs of capital are independent of the quantity of provided public good (in the short term) and concern investments realised for a key action for the provision of the public good (e.g., trail extensions or building of sanitary facilities, restoration of forest ponds). Recurrent costs

³ This section draws also on the background report “A report summarising the existing knowledge basis for assessing the direct cost of provision in the case study areas” of the NEWFOREST project <https://cordis.europa.eu/project/id/243950>

may not necessarily depend on the intensity of the operation and are therefore termed fixed operational costs while costs directly depending of the intensity is termed variable operational costs.

Opportunity costs

The opportunity cost is the benefit that can be obtained from an alternative use of the same resource. In our case, the resource is a piece of land with forest and the benefit is corresponding is the outcome of the current management. One could also consider that a given forest land could be used for agriculture or urban development. However, the main focus in NOBEL is to evaluate different forest management plans and not to consider alternatives to forest. If afforestation is considered as an ecosystem service enhancing measure, it will be relevant to consider other types of land use. The opportunity cost is the difference between the highest feasible returns that can be earned when managing the forest and the management specified in the scenario considered. It is a normal assumption that the highest feasible return correspond to the current management as it is assumed that a forest owner is managing his/her forest with respect to profit-maximization objective. However, there may be situation where a forest owner is not maximising the financial return but does in the current management account for non-economic outcomes of the forest management. Furthermore, information deficits or lack of skills may also explain why a forest owner is not maximizing profit in current management. Therefore, when estimating the opportunity costs it is important to explicit about the status quo situation. This 'opportunity cost' concept is applicable as well in the case of a trade-off between joint productions: in the case of two substitute products, any increase in the production of one product could lead to a reduction in the production of the other. It is here important to highlight the potential complex nature of opportunity costs of a management change when a change influences several services (Konrad et al., 2017; Plantinga and Wu, 2003). For example the opportunity cost of increasing carbon storage in a forest (by for example increasing the rotation age) includes the loss of timber sales but one may also have to consider that this change in management may increase the recreational value of the forest. Then, the question arises whether the opportunity cost of carbon storage should be extracted the increased value of the recreational service (if possible to monetarise the recreation value) to establish a net opportunity costs of carbon sequestration or should the opportunity costs only consider the loss in revenue by the forest owner and then note that there are co-benefits (e.g. recreational values) of increasing carbon sequestration. This question is further discussed in relation to feedback costs.

Note that the determination of the relevant opportunity cost measure is subject to feasibility of alternative land uses and hence incomes from land. Not all land uses are feasible, for practical, ecological and judicial reasons, on all pieces of land. Further, the opportunity costs are expected to vary significantly between locations and forest owners and depend on soil fertility, climate, technology (species, management systems, harvest technologies, timber markets and forest owner skills)(Juutinen et al., 2008).

Transaction costs

Transaction costs (TC) in the literature were primary addressed to a private firm context by (Coase, 1960) who defines TC as "the costs of the price mechanism". TC comprise all the expenses linked to research and information, negotiation, and monitoring where there are a contractual relationship between the provider and the consumer of the service (McCann et al., 2005). This aspect seems in

particular important related to the development of new market mechanisms. TC of PES schemes are often considered high due to issues of measurement of services provided and asymmetric information (Phan et al., 2017b). Time is frequently used as a unit of measure combined with estimated personnel costs. It may also include payment to third parties involved in organisation of meetings or monitoring of contracts. It can be measured through data from public records or through surveys from stakeholders (industries, institutions, or individuals) (Phan et al., 2017a).

Feedback costs: assessing joint-production characteristics

Feedback costs are the effects of the provision of a particular forest ecosystem service both on other forest ecosystem services (e.g. recreation or habitat provision) and on ecosystem services in other neighbouring land uses (e.g., the protection of biodiversity in forest can be potentially damaging to some culture in agricultural lands). An assessment of feedback costs require knowledge on the links between timber and non-timber benefits and potential spatial effects. In our context, the feedback costs are an indicator of joint-production characteristics: to what extend does a management scenario aiming at improving carbon storage impact biodiversity? Does investment for recreational facilities have an impact on biodiversity? For example, the cost of protecting drinking water resources against pesticides on agricultural land through afforestation may be considered a costly instrument without considering co-benefits, while accounting for carbon sequestration and recreational benefits the net protection cost of protection water resources by afforestation may be highly cost-effective (Valatin et al., 2022).

Methods for valuation of cost of provision

Two main approaches are considered. Technological approaches focus on outcomes and objective costs, while behavioural approaches focus on the forest owners' decision process and perceived costs.

Technological approaches

Bottom up engineering production approach

The traditional approach to describe the forest management is the model of (Faustmann, 1849). This approach largely used by forest engineers consists of computing the optimal rotation date by maximizing the present value (i.e., the sum of discounted net cash flow over an infinite time period) of a single-aged timber stand. (Hartman, 1976) generalizes the Faustmann model by including values associated to standing trees (which could be ecosystem service values). This leads to a change in the optimal harvest timing depending on the nature of the forest services and timber value. While the Faustman model is an economic model the input used to optimize the rotation age is based on some kind of growth model that provide link between timber (volume and quality) growth and time.

More generally, the basic idea of engineering models is that the outcome of ecosystem services are simulated using growth models/stand models/ecosystem models for alternative management scenarios (e.g. Duncker et al., 2012). These scenarios can either be predefined or can be chosen, applying some kind of (multicriteria) optimisation method. This approach is key element in the NOBEL project (e.g. Borges et al., 2014; Knoke et al., 2016). The costs of changing management (or restrictions

on the minimum supply of certain non-market ecosystem services) are estimated by multiplying the changes in inputs and outputs quantities multiplied with markets prices of these inputs and outputs. See discussion in (Rummer, 2008) in relation to fire prevention. Examples of the application of the engineering production approach include (Rosenkranz et al., 2014; Rosenkranz and Seintsch, 2015; Wüstemann et al., 2014).

Multiproduct cost function – econometric estimation

An accurate measure of non-timber outputs from current management and the total costs of production allows to estimate econometrically a cost function, representing the joint production of timber and non-timber benefits (e.g., recreation, biodiversity, carbon, water quality) (Lambini et al., 2018). The technology can be represented by an econometrically estimated production function or by a transformation function in the multiproduct case, which links produced (private and public) goods and inputs. Nevertheless the cost function, the dual representation of the technology, is following (McFadden, 1978) a “sufficient statistic” since it captures all economically relevant information about the technology.

The characteristics of timber production and of the provided non-market services can give an unusual structure of the (neoclassical) multiproduct cost function (Bowes and Krutilla 1989). Indeed, some environmental services can be produced at no variable costs as a result of natural forest growth (e.g., carbon sequestration), whereas other ones, such as biodiversity, are less compatible with industrial production of timber (including clear-cutting). To the best of our knowledge, there are only few articles on the domain of forest dealing with this methodology (Hof et al., 1985; Lambini et al., 2018). However, multifunctional agriculture offers a few studies of interest using the concept of joint production of commodities and environmental goods (e.g., Ruijs et al., 2015; Sauer and Wossink, 2012).

Forest owners’ preferences for increased amenity provision

In the forest economics literature, there are essentially two types of behaviours among private forest owners that are studied (Binkley, 1981; Helfand et al., 1994; Pattanayak et al., 2003, 2002): (1) Industrial owners manage their forests as profit-maximizers. In this case, the Fisherian separation of consumption (amenity services and others personal consumption) and production (timber) is respected. (2) Non-industrial owners produce timber and externalities, also consuming timber and other services. Their production decisions are not separated from their preferences or their utilities. Thus, their preferences (or individual utility functions) do influence the perceived costs of providing public goods. To which degree non-industrial owners value the provision of public goods and services require typically a comparison between the revealed behaviour (actual management) and the profit-maximising management (Scarpa et al., 2000). Each owner will have an optimal combination of goods and services that maximizes his or her utility which include both profit and services. This combination depends on his or her personal characteristics: preferences, income, location, bequest intention, etc. (Binkley, 1981; Pattanayak et al., 2002). The estimation of the opportunity costs would therefore be individual specific and be calculated as the loss of profit net of perceived owner benefit of an increase in non-market services.

Cost estimation with environmental evaluation methods

Methods typically used in economic valuation of non-market benefits (see section “Economic valuation of services”) have also been used for estimation of opportunity costs and include revealed preference and stated preference methods. When based on revealed preferences the opportunity costs are derived from observing the decisions of forest owners. For example, deriving the costs or the supply

curve from analysing the bids by landowners in environmental or conservation auctions (Jack et al., 2009). Hedonic pricing methods could, in principle, be used to estimate the opportunity costs of restrictions on forest management, e.g. restriction on timber production or minimum carbon storage. The basic idea is that sales prices of forest land with and without restrictions are compared and the difference representing the capitalised opportunity costs of restrictions (e.g. Ay and Latruffe, 2017; Maddison, 2000). However, this approach is mostly used to evaluate agricultural land as extensive databases on forest land sales prices are often missing. The econometric approaches to costs estimation (see section “Multiproduct cost function – econometric estimation”) are in principle also a revealed preference method as the estimation of the marginal cost of increasing the supply of services is based on observing the production and costs in existing management.

A limit of the revealed preference method is however, that it is based on observing behaviour and therefore only possible to evaluate supply of services or forest management actions that are already implemented. To evaluate new, not yet implemented measures, stated preference methods can be used and has been used to estimate the cost of provision for forest ecosystem services. With this approach the forest owners are asked to state directly their need for compensation if they should provide more services than they provide in their current management. The stated preference methods are essentially applying the contingent valuation or the choice modelling approach. In the contingent valuation method, a forest owner is asked directly their required compensation for increasing their supply of a given ecosystem service or they are asked if they accept a given contract with a specified compensation level that may vary between respondent (Gutierrez-Castillo et al., 2022). This allows to estimate forest owner’s willingness-to-accept to engage in provision of the service. The choice modelling method gives to the respondent the choice between several alternative engagements characterised by different attributes. These attributes can describe the services to provide or the specific management practices to ensure provision. Then, estimates for each attribute are derived in monetary units applying econometric methods. There are now many applications of the choice modelling approach to reveal forest owners willingness-to-accept increased provision of biodiversity protection, recreation opportunities, climate change adaptation, or carbon sequestration (see e.g. Abildtrup et al., 2021; Gadaud and Rambonilaza, 2010; Mostegl et al., 2019; Rabotyagov and Lin, 2013; Vedel et al., 2015, 2014).

Economic valuation of services

In this section we will discuss the relevance of different methods to estimate the value of non-market good and services from forest. We first discuss the main preference based methods, i.e. revealed and stated preference methods and then also considering non-preference based methods, e.g. alternative cost methods.

Revealed preference values

Market based methods

A straight forward approach is to use market prices. If we assume that the timber market is well-functioning we can use the timber price as a measure of the social value of timber production. In this case, we, per definition, do not talk about non-market valuation, as the market exists. However, there may be other services where there exist no direct market price but there are related markets. For

example, one can use the market prices of mushrooms to value the quantity of mushroom collected for household consumption.

One can also observe prices from simulated markets or market experiments (Chakrabarti et al., 2019). An example is the crowdfunding platform ecosel (www.pay4ecosystem.eu) that allow to estimate consumers willingness to pay through their real donations (Tóth et al., 2013).

Revealed preference methods

The most common revealed preference method used to evaluate the FES is the **travel cost method** which have been widely used to estimate the WTP for forest recreation opportunities (e.g. Bartczak et al., 2008; Borzykowski et al., 2017a). This method derives the value from observing the current use of forest for recreation, including distance travelled and number of visits to a forest or to different forests (Parsons, 2017). In the example section we will show the application of the so-called site selection model (Bockstael et al., 1989) which is estimated with the random utility model (McFadden, 1977). This method allows explicitly to account for forest characteristics and the substitution between forests as recreational sites. This make the model useful to estimate changes in management and marginal changes. Another method, which have often been used to evaluate the amenity values of forests is the **hedonic pricing method**. Based on observed transaction prices of houses the WTP for being in the proximity of forests can be derived. However, the estimated values are typically an aggregate of different services (recreation value, air purification, landscape values, etc) and are often not linked to the specific management of the forest (Tuffery, 2017) and therefore less appropriate for assessing the value of specific services related to specific management. It is also possible to evaluate a non-market service by revealing its value in producing other services, i.e. **the production function approach** (Bockstael and McConnell, 2007 chapter 8 and 9; Johnston, 2022). This has for example been used to estimate the water protection service forest by analysing the effect of water quality on the cost of drinking water provision (Abildtrup et al., 2013a).

Stated preference values

Stated preference methods represent a different approach that is based directly on the declarations by the consumers. More specifically, this includes methods like contingent valuation or choice experiments (Johnston et al., 2017). This involves typically a survey of a sample of the population considered, where scenarios for changes in the supply of ecosystem services are described and the respondents are asked to provide their WTP directly (contingent valuation) or the WTP is derived from their choice between different hypothetical scenarios which involve some kind of payment (choice experiments). A crucial assumption in such methods is that it is possible to present the scenarios in an understandable way for non-specialists, which is a challenge (Johnston, 2022). Therefore, applied studies of the WTP for non-timber services have more often focused on the WTP for alternative management options and less directly on indicators of ES provision. A recent review of economic valuation relevant for forest management is carried out by (Müller et al., 2019).

Cost based methods

Estimation of the economic value of non-timber values can also be estimated using so-called *cost-based methods*. These are normally not considered economic valuation methods as the value is not based on consumers' preferences but on, for example, the lowest possible alternative cost of providing the same service in another way or the cost of restoring of a degraded ecosystem. However, it is the

method typically used when estimating the value of carbon sequestration. The social value of reduced carbon sequestration is extremely difficult to estimate, as it is a potential benefit for the whole world population. However, based on national targets and policies it is possible to derive what the alternative cost of emission would be in the cheapest alternative. In practice, governments often provide a reference value for the carbon emission reductions.

There are different variations of the cost-based or substitution methods (Brouwer et al., 2013; Brown, 2017; Masiero et al., 2019). **Replacement** or **restoration** cost approaches estimate the cost of re-establishing a service if it has been removed. This could be the cost of establishing a new forest which correspond to a forest that has been removed. **Damage cost avoided** estimate the cost of damage avoided by the ecosystem service, e.g. the loss of agricultural production if the pollination service was not available or the cost of purification of water at treatment plant if forest is protecting the water quality.

Benefit transfer

Benefit transfer are often considered as a more rapid and less costly approach to obtain economic values. The idea is that pre-existing empirical estimates from one or more settings where research has been conducted previously to predict measures of economic value or related information for other settings (Johnston et al., 2021; Johnston and Rosenberger, 2010). There are different methodological approaches to benefit transfer. The two main groups of methods are *unit value transfers* and *benefit function transfers*. Unit value transfer is relatively simpler as values from a similar case or a set of similar sets of cases are used to approximate. The main the value in the case of interest. The challenge is to identify if there are similar cases and if there more studies exist, then to choose which of them to use. In the benefit transfer function approach a parametric benefit transfer function is estimated based on existing studies. The value of the benefit is described as function of the characteristic of the projects evaluated, the environmental and sociodemographic contexts. Often the benefit function is estimated through a meta-analyses, that systematically exploit existing studies, applying statistical methods (Grammatikopoulou et al., 2021; Johnston and Bauer, 2020). A recent guide of the different steps and choice to make in a benefit transfer study can be found in (Johnston et al., 2021).

Elsasser et al. (2021) shows an example of using benefit estimates from other studies to map the economic values of ecosystem services in Germany while Grammatikopoulou and Vackarova (2021) estimate the forest ecosystem values in a meta-analysis on a European scale and apply the results to estimate the ecosystem values of Czeck forests. There are also increasing number of ecosystem services value mappings where the values are obtained from primary valuation studies (Schägner et al., 2013), typically corresponding to a functional benefit transfer approach.

There exist a number of databased for primary studies including the Environmental Valuation Reference Inventory, EVRI (www.evri.ca), and also some specific forest focused databases (Elsasser et al., 2016, 2009; Stenger et al., 2009).

However, Müller et al. (2019) review available ecosystem services value estimate with the purpose to assess they usefulness as input for forest management and policy decision-making. They find that existing value estimates vary significantly and depend heavily on the forest context and conclude “... *integrating forest ES values in decision-making and planning processes using the BT of existing data*

seems difficult". Therefore, local valuation studies are necessary if values should be integrated in decision support tools.

4. Evaluating market mechanisms

A central point of the NOBEL projects is to evaluate the performance of alternative markets mechanisms. This evaluation will have several aspects. First we have to know to which degree the mechanism lead to a social optimal level of provision. While governmental PES schemes may include specific provision target – potentially based on a cost-benefit analysis, the provision level is not controlled with crowd-funding mechanisms. Therefore, such mechanisms should also be valued with respect to their ability to lead to social optimal solutions. As the majority of the non-market services are public goods, freeriding is an issue to address in crowdfunding. Several studies has considered the relationship between willingness to pay under voluntary (e.g. crowdfunding) and non-voluntary schemes (potentially incentive compatible) (Bouma and Koetse, 2019; Roesch-McNally and Rabotyagov, 2016; Swallow et al., 2018) or have explicitly investigated specific mechanisms like auctions (Liu and Swallow, 2019), and the Ecosel platform (Rabotyagov et al., 2013; Roesch-McNally et al., 2016). Boume and Koetse (2019) conclude research is needed to reveal how context and mechanism designs influence voluntary donations.

One aspect influencing the efficiency (and effectiveness) is the flexibility of the mechanism to account for spatial and forest owner heterogeneity, e.g. by spatial targeting (Wunder et al., 2018). Another aspect is the level of the of transactions costs as in PES schemes they represent a significant share of the costs (Abildtrup et al., 2012; Wang et al., 2017).

5. Examples

In this section we will discuss how a number of key services considered in the NOBEL project can be evaluated. As starting point we will use the French pilot study case as an example. An recent example valuating benefits of forest ecosystem services can be found in (Zanchi and Brady, 2019).

In the following section relevant methods by ecosystem services valuation for the French pilot is considered. However, it should be noted that local conditions, study objectives, and time and resource constraint will influence what is the best strategy in given region. As example, we use the French pilot demonstration case (Table 3). Valuation of climate change mitigation and recreation will be treated in more details as relevant for the example forest stands considered in the NOBEL project.

Table 3. Overview of economic valuation approaches relevant for the French pilot demonstration region.

Ecosystem Service	Cost assessment	Unit	Economic value	Present value of Benefit and costs
Climate change mitigation (Carbon emission reductions)	Opportunity costs based on engineering models (current profit minus profit in alternative management): Using output from growth models and standard unit costs and unit prices on timber	Tonnes C/ha/year	Alternative costs approach – use governmental validated values, e.g. “valeur tutélaire du carbon” . Value per unit of carbon	Discount future costs and benefits
Providing habitats	Opportunity costs based on engineering models (current profit minus profit in alternative management): Using output from growth models and standard unit costs and unit prices on timber	Size of areas not harvested, tree species diversity, amount of deadwood, tree size diversity.	Use value: use recreational values estimated as a function of different habitat indicators or proxies: deadwood, tree species diversity, and tree size diversity in Abildtrup et al. (2020) ⁴ . Non-use value: use primary survey results based on economic non-market valuation methods. Compare with values from similar studies	Annual values
Recreation	Cost estimations of maintaining visitor facilities, e.g. waste management, signs, impact on hunting values	Visitors/ha/year	Use forest recreation model (Abildtrup et al. 2020) to estimate number of visits and WTP (Maximum entrance fee) in the specific forests considered in the	Annual values

⁴ Abildtrup, J., Garcia, S., Kervinio, Y., Sullice, E., Tardieu, L., Montagne-Huck C. 2020. [Les usages récréatifs des forêts métropolitaines Un état des lieux des pratiques et des enjeux](#). FESE, Ministère de la Transition Ecologique

				pilot demonstration case	
Water protection/purification - relative to other land uses ⁵	Difference in Net present value of revenues from forest and alternative more profitable land uses	Quantity of ground surface water generated per ha, Nitrate and pesticide concentration in water generated	of Drinking water or value is calculated as the quantity of water times the differences in cost of provision of water as a function of water quality (Abildtrup et al., 2013b)		Annual values

Climate regulation: Carbon sequestration and substitution

Here we will focus on the situation where changes in management will lead to change in carbon emission either through changes in sequestration or substitution of fossil energy. Often the role of the forest for mitigating climate change is related to measures to hinder deforestation or to afforestation measures, including short rotation managed to produce biomass (van Kooten and Johnston, 2016), but this is not in the current French context the most relevant measures.

One of the first questions to consider when evaluating the cost and benefit for changing the provision of climate change mitigation services is the question if we only will consider the carbon sequestration in the forest or we will also include carbon sequestration in the timber products as well substitution effects. The substitution effects are reductions of the emissions due to reduction in the use of fossil energy as a result of substituting fossil (based products) energy with wood products. The answer to this question will, to high degree, be determined by the objective of the estimation. If it is from the forest owner perspective with the objective to sell carbon credits the quantification will be based on the credit specifications. For example, the projects labelled with the Label Bas Carbon⁶ in France include both storage in timber products and substitution effects and it is specified how sequestration in forest products and substitution effects should be accounted for to be eligible to a certification. If the perspective is the national government, then it will be more important to consider the international rules for carbon accounting related to complying with national carbon emission reductions targets. Based on the answer to this principle question, the quantity has to be estimated, including the time profile of reductions. The latter can be important as current reductions are often considered more important than reductions which take place in the future. At the same time, as discussed below, the price of carbon is often considered to be increasing over time. Modelling and measurement of the impact of changes in forest management is not the objective of the present report. Here will focus on how to estimate the costs of one ton of carbon emission reduction.

There are two main approaches to estimate the value of carbon emission reduction. Estimates based on the social value of carbon reduction and an opportunity cost method. The social designates the economic cost caused by an additional ton of carbon dioxide emissions or its equivalent, or more precisely, the change in the discounted value of economic welfare from an additional unit of CO₂-

⁵ Having operational models which can quantify the link between forest management and water quality and quantify provision

⁶ JORF 2018 Décret No 2018–1043 du 28 Novembre 2018 créant un label “Bas-Carbone”, Paris

equivalent emissions (Nordhaus, 2017). However, this value is often controversial as estimated under a number of assumptions, e.g. about future technological development, which can be discussed. As most countries have national target for reductions in emission and these targets are not directly linked to the social value of carbon reductions but results of international negotiations, it makes sense from a national perspective to use the opportunity cost of carbon which corresponds to the cheapest alternative measures to reduce carbon emissions. In other words, at what cost can the sequestration of one additional ton of carbon in the forest be replaced by the least alternative expensive technology? Typically, government will propose “carbon prices” which may be time dependent.

In France the government publishes reports guiding the valuation of public investment projects (Quinet, 2019, 2013) and it has been suggested to use these for valuation of climate mitigation services (Therond et al., 2017). Quinet (2019) propose a price of carbon emission of 87 Euros/tCO₂e in 2020 and increasing linearly to 250 Euros/tCO₂e in 2030, and then linearly increase to 500 Euros/tCO₂e in 2040 and using Hotelling rules to 2050 and obtaining a value of 776 Euros/tCO₂e. These values are based on target for France in the Paris agreement (max +1.5 °C).

In a first attempt to value forest owners proposed projects for given forest stands we will discuss the forest stand A and B in Table 4. The parcel is located south of Nancy in the *departement* Meurthe-et-Moselle (Figure xx). These examples are based on forest stands where forest owners have proposed changes in management to increase provision of ecosystem services. The management changes considered are extensions of the rotation age and conversion to uneven-aged management.

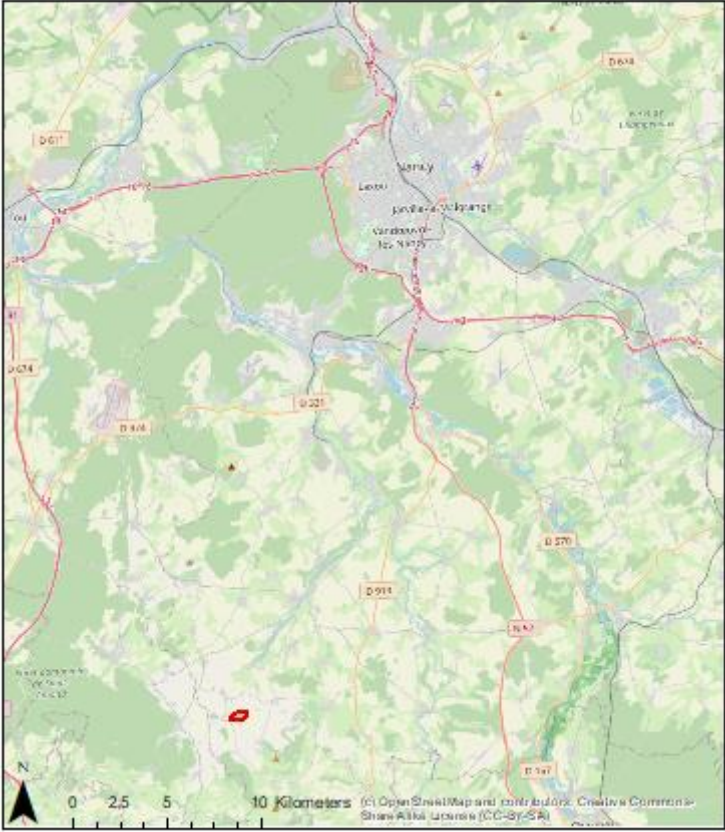


Figure 1 Location of forest with forest stand A and B in Table 4 (red borderline).

Table 4: Composition, ages and possible alternative managements for the four stands studied (Leflon, 2020)

	Species	Age	Area	Change suggested
A	Sessile Oak (60%),	115 years	1.87	Increase rotation age from 150 to 180 for oak and from 85 to 100 for pine.
	Laricio Pine (40%)	70 years		
B	Sessile Oak (30%),	115 years	3.0	Transition to uneven age management.
	Laricio Pine (70%)	70 years		

We carry out the analyses with two different assumptions about the time perspective. In the first analysis we consider a full rotation period (a long term effect) and in the second we consider a 30 year time horizon that corresponds to the certification rules of the LBC. Furthermore, the rules for calculating the carbon sequestration in the forest and in the timber products and the substitution effects are based on the guidelines of the Label Bas Carbon. This implies, among other, that the timber production is modelled based on production tables or simple growth models, and the produced timber is multiplied by tree species-specific conversion factors that allow estimation of Carbon storage in the forest and in the timber products and estimation of the substitution effects. To estimate the opportunity costs we used prices of timber products and discount rates applied by the forest owner. Table 5 shows the annualized economic return for the two forest stands considered as well as the carbon impact for one rotation. As the rotation period is different before and after the implementation of the measure we calculate the annual values. Furthermore, it should be noted that when we consider the average effect over a period we assume that after a final harvest the net change in carbon sequestered in the forest is zero. Therefore, only the storage in forest products and substitution counts and as the increase in rotation age decrease the average annual production the change in management has negative effect on carbon emissions (as average annual production is lower). However, this ignores that the carbon the average storage of carbon in the forest will be higher in the suggested alternative scenario.

Table 5: Income and average carbon stored per year on one rotation for stands A and B for both scenarios (Leflon, 2020)

	Income for mixture A (€/ha/year)	Income for mixture B (€/ha/year)	Average CO ₂ stored for stand A (tCO ₂ /ha/year)	Average CO ₂ stored for stand B (tCO ₂ /ha/year)
Usual	16.59	14.43	4.95	6.52
Suggested	13.96	13.90	4.66	6.20

However, the time profile of the net emissions is important. In figure 1 we have included the impact of the scenarios during a 30 years period to calculate the average reduction per ha and per year corresponding to the time period considered in the certification *Label Bas Carbon*. We see that the sequestration in the forest over the next 30 years (the change in carbon stored between the initial situation and the end of the 30 years period) will be lower with new management regime as the final harvest takes place later (Pine, stand A) and therefore the new regenerated stand will be younger at the end of the period. On the other hand, the stored amount of carbon in final products and substitution increases with the change in management for the 30 years period. In total the reduction in CO₂ over the 30 years per ha is 64 tCO₂ and 75 tCO₂ for forest stand A and B, respectively. Assuming

the CO₂ prices corresponding to (Quinet, 2019) and a discount rate of 4.5% we arrive at a present value of the carbon storage to be 11,455 Euros per ha and per 30 years for stand A and 13,282 Euros per ha and per 30 years for stand B. Note that we here used the carbon prices from a policy perspective. Whether this price can be obtained on carbon credit market is not given. Brèteau-Amores et al. (2022) report values of 5-50 euros/tCO₂ in realized LBC projects. From the forest owner perspective, the opportunity cost of the changes in management should not only consider the 30 years period but include the impact on the forest after the 30 years. As an approximation we can use the difference in the annualized value from one rotation before and after change in management. According to table 5, this corresponds to an opportunity cost of 2.63 Euros/ha/year for forest stand B and 0.53 Euros/ha/year for forest stand B and with a present value of 45 and 9 euros/ha/year corresponding to 30 years time horizon. This low opportunity cost should be seen in the context of the very low income from the current management.

This case shows the importance of considering the time period considered. Often when it concerns carbon credits the impact on carbon emissions will concern a limited time period and this may raise important questions about how to account for the carbon effects. We show also that the question of including sequestration in forest products and substitution of fossil energy can influence the results significantly. Sequestration in forest products and substitution have, for example, been ignored in (Zanchi and Brady, 2019). Ideally, the carbon stored in belowground should also be included in the analysis as in Brèteau-Amores et al. (2022) but was ignored in Leflon (2020).

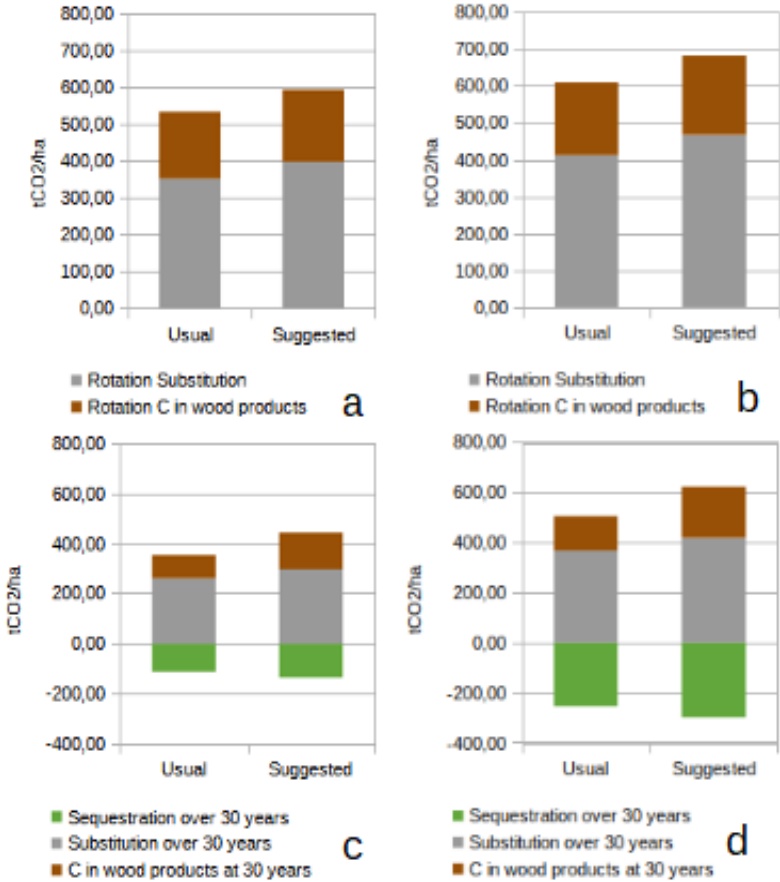


Figure 2: Evolution in the amount of carbon of the different compartments predicted by the LBC for stand A on one rotation (a), for stand B on one rotation (b), for Stand A on 30 years (c), and for stand B on 30 years (d) (Leflon, 2020).

Forest recreation

In the literature there are several studies that has estimated the economic value of forest for recreation use (Abildtrup et al., 2020; Borzykowski et al., 2017b; Garcia and Jacob, 2009; Zandersen and Tol, 2009). However, these studies estimate average values for current state of the forests and are less applicable to estimate changes in recreational quality. However, there are examples of studies accounting for recreational quality of forest. This is mainly so-called site selecting models using the random utility model (Abildtrup et al., 2020; Termansen et al., 2013). Such studies allow to assess changes in recreation values as a function of forest characteristics for a given specific forest on a specific location. A disadvantage of this approach is, however, that often the characteristics of forests accounted for in the model are not those impacted of changes in forest management. For example, the age of trees or if the forest is managed wit even or uneven aged forest stands are often ignored due to lacking GIS maps of tree ages (or height) which are relevant when evaluating the effect of increasing the rotation length or changing management type on the recreational value of a forest. Information about the population's preferences on tree heights and stand management have been estimated in stated preference studies (Abildtrup et al., 2020) but such studies does not allow to estimate how many visitors in a specific forest.

Here we will approximate the change in recreational value per year and ha of changing management from even-aged to uneven-aged management combining stated preference studies with a site selection model using results from Abildtrup et al. (2020) which is based on three surveys of the French population's use of forest 2017-2018. Figure 2 shows the average number of annual visits per ha in forest in Meurthe-et-Moselle simulated using the estimated random utility model using the survey data. Obviously, the highest number of visits are found close to the main urban agglomerations (Nancy and Thionville). The average number of visits per year in the forest stands considered here (indicated by the red circle in the Figure 2) is 12 visits per hectare per year. This low number of visits is due to low population density and high forest density in the area. Furthermore, it is a small private forest. The analyses show that such forests are relatively less attractive for visitors. The results of the choice experiment showed that a visitor would accept to go 9.2 km longer (18.4 km two-way distance) to visit a forest with an uneven-aged forest management. However, this value is estimated in the assumption that there is no alternative forest with un-even-aged management that could be visited instead of this forest. Therefore, this estimate is a maximum estimate. If we use a travel cost of 0.36 euros/km which include opportunity cost of time of an average visitor (Abildtrup et al., 2020 p133) and 12 visitors per hectare we have that the annual increase of the recreational value of stand B will be $3 \text{ ha} * 12 \text{ visits/ha} * 18.4 \text{ km/visit} * 0.36 \text{ Euros/km} = 238 \text{ Euros}$. However, deciding to convert from even-aged management to uneven-aged management will take time. Therefore, this estimated annual value is assumed only to be obtained after 30 years. Assuming as linear increase of the recreation value and a discount rate of 4.5% lead to a value of 97 Euros per year for the forest stand B. It is here important to highlight that this is a maximum value as assuming no alternative forests with uneven-aged management and also that we account for opportunity cost of time in the travel cost estimation. It could be argued that opportunity cost of time should not be included as the travel time to the forest may not be seen as a cost for the visitor (Abildtrup et al., 2020).

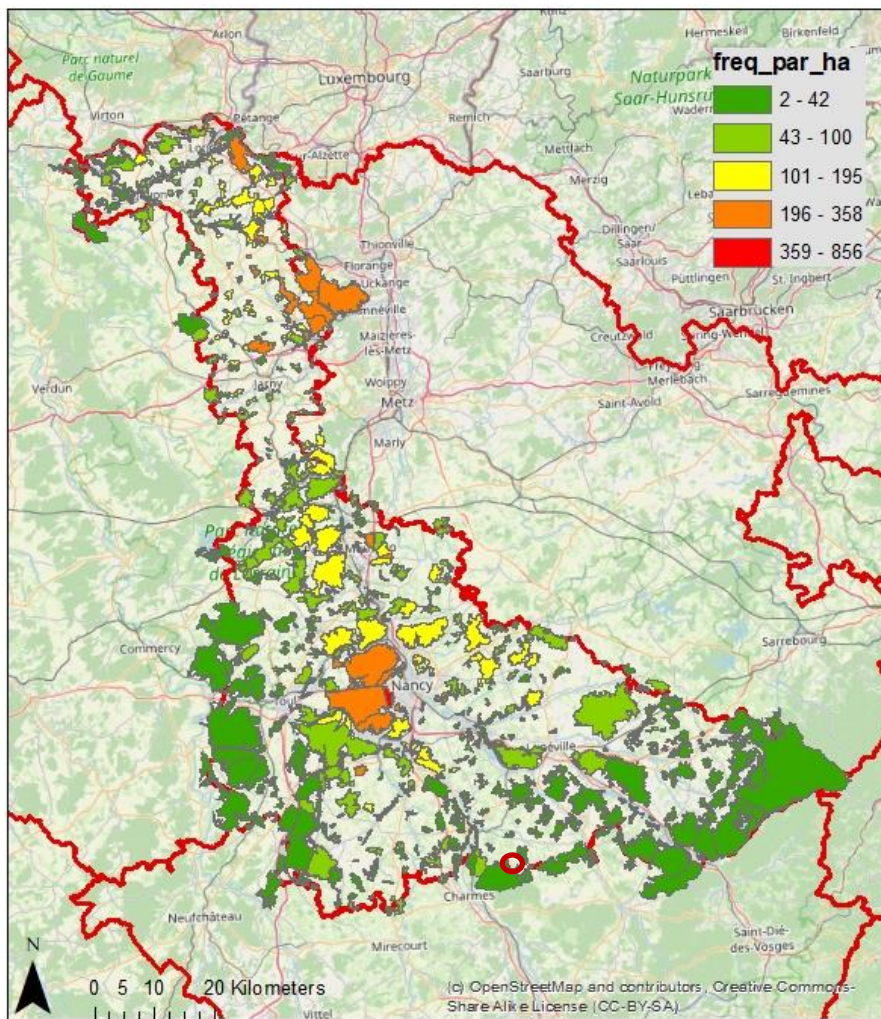


Figure 2 Average number of visit per per year and ha in forests larger than 25 ha in Meurthe-et-Moselle. The case forest is indicated in a red circle.

6. Perspectives

Forests are experiencing an increasing demand for offsetting carbon emissions by individuals and companies. At the same time there is an increasing demand for biodiversity protection and, in particular, it is demanded by companies supporting forest-carbon projects that such projects will, as a minimum, not have a negative effect on biodiversity conservation. This raises a strong demand for documenting not only the carbon effect of forest projects but also to document the impact on biodiversity. However, in this report the focus was not on the definition of ecosystem service indicators and how to measure these. To facilitate the emerging market for ecosystem services it is important that there is more research into operational procedures for measuring ecosystem service provision that also account for uncertainty and the long time horizons of forest projects.

Furthermore, for companies it is, generally, important to document that the projects supported are *additional*, i.e. the projects would not be implemented without support. A key element in assessing the additionality is to estimate the opportunity costs associated with the change of management. If there are positive opportunity costs of changing management we will expect that supporting the

project will be additional. However, for a private non-industrial owner the estimated costs should also account for the owners' private utility of the non-marketed ecosystem services, e.g. an increased recreational. However, as this private utility is subjective it is difficult to include in the cost assessment and to document. There is a need for further research into how should forest owners' private utility of ecosystem services be addressed in cost assessments. Furthermore, research in the NOBEL project indicate that individuals having a willingness-to-pay to forest projects are less concerned about additionality as long as project support a forest management that are sustainable (Frings et al., 2022).

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