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Sources and types of uncertainties in the information on forest-related ecosystem services

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ABSTRACT

The concept of ecosystem services has gained importance in the forest management and forest policy processes in recent years. Ensuring the sustainable provision of ecosystem services requires accurate information of the current provision and methods for predicting the impact of important drivers, such as changes in land cover and land use. In this review, we define the sources of uncertainties in forest-related ecosystem service assessments and discuss their importance to the usability of the information for different purposes. The uncertainties are due to e.g. variation in the selected indicators for the ecosystem services, lack of primary information on them, poor correlation with the data used for mapping the ecosystem services to larger scale and for predicting the impacts of human interventions. The uncertainties can be random or non-random and their assessment is often ignored, especially in the case of the non-random errors. As a result, different assessments and subsequent decision recommendations can be highly conflicting. We do not expect that the accuracies would significantly improve in the short term. The best way to proceed is therefore to assess the uncertainties and take them into account in the decision making for forest management.

1. Ecosystem services concept as a means to promote sustainable forest management

Ecosystem services (ES) represent the goods and services derived from the functions of ecosystems utilized by the humanity (Costanza et al., 1997, 2017; Crossman et al., 2013). The concept of ecosystem services was originally designed as an educational and communication tool (Daily, 1997) to acknowledge that human wellbeing is tightly connected to the provision of these services. Nowadays the concept of ecosystems services is the main framework for environmental policies and monitoring (Norgaard, 2010). The European Commission emphasizes the importance of accurate information on ecosystem services as the basis of the EU Biodiversity Strategy for 2020 (European Commission, 2011). Land use change is the most important driver affecting the ecosystems (Dong et al., 2015). As a result, ecosystem services have been emphasized in national and regional land use policies and planning (e.g. Frank et al., 2015, Haakana et al., 2017, Tammi et al., 2017). Policy makers are increasingly recognizing the potential of ecosystem service mapping in strategic planning (Vorstius and Spray, 2015).

In the cascade model (Fig. 1), the ecosystem services are addressed through the structure and process of the ecosystems and their functioning, benefits and value obtained from the used ecosystem services. The biophysical structures and processes create the basis for the functioning of the ecosystem and the functions create the capacity to provide services. The capacity to deliver a service exists independently of whether anyone wants or needs that service, but that capacity becomes a service only if a beneficiary can be clearly identified. The value of the benefit can be defined as economic, social, health or intrinsic value (Haines-Young and Potchin, 2010). The ecosystem services approach has been criticized, however, for taking a fully anthropocentric view and hiding the intrinsic values of nature (Fürst, 2015).

Ecosystem services can be grouped in many different ways (e.g. de Groot et al., 2002, MA, 2005; TEEB, 2010). In Common International Classification of Ecosystem Services (CICES), which is used in this review, the ecosystem services are divided to **provisioning**, **regulation and maintenance**, and **cultural** (Haines-Young and Potchin, 2010). Services that are most relevant from forest management point of view include provisioning services such as timber, berries and mushrooms, game, reindeer, and bioenergy; regulating and maintenance services such as climate regulation; and cultural services such as recreation and nature tourism. In the following text, we use the term ‘ecosystem services’ to refer to all possible forest-related ecosystem services in general and differentiate between them only when it is relevant from the point of view of data acquisition. In those cases, we always spell out the specific ecosystem services or steps of the cascade model (Fig. 1) we are

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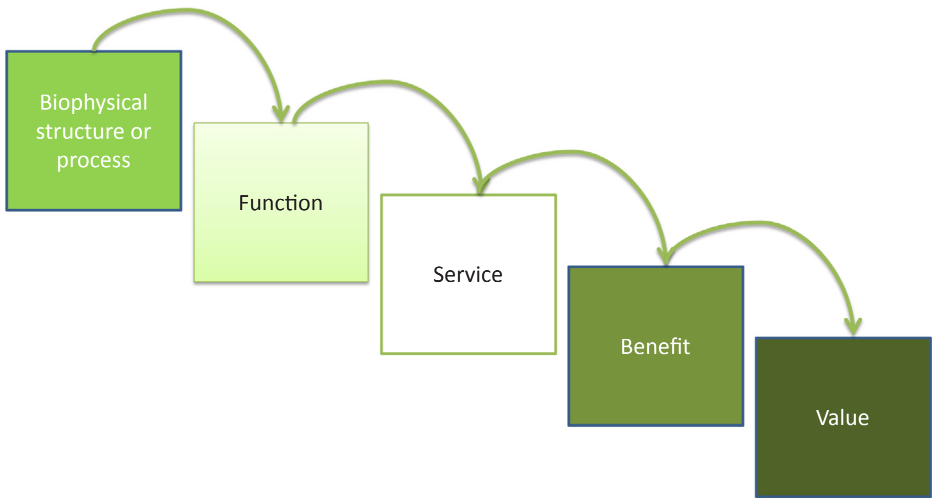

Fig. 1. The cascade model (modified from Haines-Young and Potchin, 2010).

referring to.

The ecosystem services are operationalized through a selected set of indicators (e.g. Müller and Burkhard, 2012). The purpose of the indicators is to support the management of ecosystems and to communicate on their condition. Thus, they simplify the complexity of the ecosystems to manageable concepts. The set of relevant ecosystem services and their indicators varies from region to another. For instance, Mononen et al. (2016) and Hansen and Malmaeus (2016) have presented a different set of indicators for Finland and Sweden, respectively, even though the two countries resemble each other very closely in terms of forest structure. This variation is one of the challenges when comparing international ecosystem service assessments (Maes et al., 2012b, Mononen et al., 2016). One possible reason for the variation is that the values of the experts who carry out the selection of the criteria implicitly reflect to the selection of the indicators (Menzel and Teng, 2010).

Sustainable management of natural resources can be seen as maximizing the social welfare obtainable from them (Kant and Lee, 2004). Sustainability means that the future generations can consume the ecosystem services to the same extent as the current one (e.g. Norgaard, 2010). Sustainable provision of ecosystem services thus requires a nondeclining provision of all services over an infinite period in time. Only changes that are inarguably sustainable are Pareto improvements, where the supply of ecosystem services improves with respect to one or more indicators but does not deteriorate with respect to any of the other services. Trade-offs are inevitably related to all other changes in the current and future provision of the ecosystem services. The decision on whether such changes are sustainable or not depends on the values of the humans making the evaluation (e.g. Fürst et al., 2010, Vorstius and Spray, 2015, Hartikainen et al., 2016). Including ecosystem services into decision making is one way to strive for sustainable forest management (e.g. Martinez-Harms et al., 2015). According to Meyer and Schulz (2017), however, forests are currently underrepresented in the studies related to ecosystem services.

Ensuring that ecosystem services are provided sustainably requires information of the current state. Sample-based information is adequate to make decisions on sustainability at national and regional scales. For decision concerning locations, such as where it is important to protect, restore or improve ecosystems or their services, a map – i.e. spatially explicit information – is required. The maps can be used, for instance, to detect hotspots or coldspots, i.e. areas with high or low supply of ecosystem services (e.g. Pagella and Sinclair, 2014). Co-occurrence of different ecosystem services in an area implies synergies and tradeoffs (Maes et al., 2012a). The maps can also be used to detect areas where the supply of ecosystem services decreases or increases due to changes in land use (Pagella and Sinclair, 2014); to identify providing and benefiting areas (Syrbe and Waltz, 2012); and to communicate the effects of policies to the land use and ecosystem service provision (Vorstius and Spray, 2015).

Real policy decisions require two or more decision options to choose from and predictions of the future consequences of these options (Corona, 2016). To ensure sustainable provision of ecosystem services, information needs to be available and of sufficient quality. We also need to have decision support tools for predicting the future development of the services affected by the decisions executed.

We review the acquisition of primary data (Section 2) and mapping of ecosystem services (Section 3), concentrating on those services relevant from forest management point of view. We review the methodology available to assess the uncertainty in the ecosystem services data (Section 4) and note that in most cases uncertainty assessment is lacking or inadequate. The search of references was carried out using Web of science on 15 November 2016. We used one keyword describing the uncertainty assessment (e.g. “error” , “uncertainty”, “validation”, “evaluation”), one keyword describing the data collection and usage (e.g. “mapping”, “inventory”, “data acquisition”) and as the last keyword “ecosystem services”. While the inclusion of all resulting articles to this review is by no means exhaustive (Web of science gave 22 652 hits for the keyword “ecosystem services”), we specifically attempted to focus on articles that acknowledged uncertainties. Finally, we discuss our findings on the gap between the information demand and supply in terms of contents, scale, accuracy and uncertainty assessment with respect to decision making.

2. Acquiring ecosystem services data

2.1. Indicators for ecosystem services

The data acquisition for ecosystem services is operationalized through a set of indicators that can be assessed. Primary data are needed for the indicators of the structure, function, benefits, and value in the cascade model (Fig. 1). For instance, the habitat area (ha), production (kg/ha/A), yield (kg), and monetary value (€) could serve as the indicators of structure, function, benefits, and value, respectively, if forest berries and mushrooms were considered as an example service (Mononen et al., 2016).

Selecting a good set of indicators is important, as those vary in quality for decision making. Auvinen et al. (2007) evaluated indicators of biodiversity using several criteria: relevance, impact, effectiveness, cost-effectiveness, acceptability, incentive value, transparency and opportunities for participation, equity, flexibility, predictability, and

Table 1
Data acquisition for the different levels of the cascade model of ecosystem services.

Method	Steps of the cascade model involved	Advantages	Disadvantages
Field data	- Structure - Function (- Use)	- Objective (unbiased) information - Uncertainty assessments available	- Costly - Laborious and impractical for some indicators - Unobservable for some indicators
Models	- Structure - Function (- Use)	- Easy to utilize in mapping the ecosystem services - Cheaper than primary data - Uncertainty assessments possible - Possible to assess the impacts of changes	- Requires primary data for estimation
Survey data	- Use - Benefit - Value	- Uncertainty assessments possible	- Prone to subjectivity - Regional averages difficult to utilize in mapping
Benefit transfer	- Benefit - Value	- Cheaper than original valuation studies	- Prone to bias due to differences between regions and context - Validation and uncertainty assessment difficult
Expert judgment	- Structure - Function - Use - Benefit - Value	- Easy to utilize in mapping the ecosystem services - Cheaper than primary data - Possible to assess the impacts of changes	- Highly subjective - Prone to bias - Validation and uncertainty assessment difficult

permanence. Forest inventory experts defined 41 variables measurable at field that could be used as indicators of biodiversity (Chirici et al., 2012). These indicators were ranked from low to high according to their feasibility and importance and those that had both high feasibility and importance were found to be the most useful.

Primary data for many forest related indicators of structure and function can be obtained from field plots of forest inventory (Table 1). If primary data are available, those can be used to form (statistical or process-based) models. Models are of importance, as they can be used for mapping the services and especially for assessing the impact of future changes. Primary data for benefits and value require surveys involving the human beneficiaries. These data can also be presented as a (benefit transfer) model. As it is often difficult to obtain the required data, most of the services lack primary information (Eigenbrod et al., 2010). The only option in that case is to assess the ecosystem services by means of expert judgment, even though such data are highly subjective and difficult to validate (Table 1).

2.2. Field data

In many countries a sample-based national forest inventory (NFI) has been established for monitoring forests (Chirici et al., 2011). The main benefit of the sample-based data acquisition is the possibility to assess the uncertainty included (Table 1). The NFIs have traditionally measured indicators suitable for timber production. For instance, the NFIs provide estimates of the area of managed forests (ha) and the increment of growing stock (m^3/ha), which were identified as indicators of wood and bioenergy provision (Mononen et al., 2016). Carbon-related indicators are also readily available, because those can be calculated from the growing stock information.

Currently, the NFIs also measure other than timber-related indicators, such as those related to deadwood, vertical structure, important habitats, and tree species mixtures. For instance, the global Forest Resources Assessment (FRA) strives for collecting information on the provision of ecosystem services globally (Miura et al., 2015). However, obtaining primary information on ecosystem services other than timber is still a challenge. Sample-based inventory data are accurate and, in principle, could include many other indicators of structure and non-wood production. In practice, the collection of such data may be resource intensive (Maes et al., 2012b) and therefore practically infeasible. A smaller sample tailored specifically for the given ecosystem service(s) is often needed to provide adequate statistics concerning, for instance, the production of forest berries or game habitats (e.g. Kurki et al., 2000, Turtiainen et al., 2011, Melin et al., 2016).

To enhance the availability of primary data, many different environmental monitoring schemes have been and are being developed (see Geijzendorffer and Roche, 2013 for a review). Most of them are related to a subset of possible ecosystem services. Several monitoring schemes have been developed for habitat or species diversity (e.g. Ståhl et al., 2011, Corona et al., 2011, Chirici et al., 2012), while the political focus of environmental protection has recently been based more on maintaining or restoring ecosystem services (Geijzendorffer and Roche, 2013).

The ecosystems provide services in different scales, which need to be accounted for also in the monitoring schemes (Geijzendorffer and Roche, 2013). The scales can be local, regional, global, or related to directional flow or distance (e.g. Costanza, 2008). Each indicator should obviously be monitored on the relevant spatial scale. Some indicators like landscape patterns require observations from wider geographical extents than sample plots. The patterns can be described using landscape metrics such as the number of patches, mean patch size, or shape and diversity indices (Frank et al., 2012; Ramezani and Ramezani, 2015). Another similar example requiring landscape measures are the recreational and aesthetic values (Pukkala et al., 1995; Frank et al., 2012).

2.3. Survey data

The benefits and value, i.e. the uptake of services, are typically difficult to assess in the field. Some indicators, such as the degree of grazing by game, may be monitored with sample-based inventories, but surveys or interviews of the beneficiaries are often needed. If it was possible to interview the beneficiaries living close to the field plots, the surveys could be directly linked with the field measurements, but usually the surveys need to be carried out as a separate effort.

Trade statistics can also provide useful information. For the provision of most of the ecosystem services, relevant statistical information exists at the national level. For instance, the amount and value of berries, mushrooms or other non-wood forest products having monetary value are assessed in several countries. The Food and Agriculture Organization of the United Nations (FAO), Center for International Forestry Research (CIFOR), International Forestry Resources and Institutions Research Network (IFRI) and World Bank have even published good practice guidance for collecting socio-economic data with household surveys (FAO, CIFOR, IFRI and World Bank, 2016).

Specific valuation studies are needed for ecosystem services which do not have market value. In general, valuating non-market goods can be carried out using either preferences revealed in real decision situations, or those stated in hypothetical decision contexts (Louviere et al., 2000). In many cases, however, there are no other possibilities except relying on the stated preferences. The stated preferences methods include e.g. contingent valuation or choice modeling (Louviere et al., 2000).

2.4. Benefit transfer data

Most of the evaluations are case studies, which involve one or a couple of services in one region (e.g. Horne et al., 2005; Japelj et al., 2016). When the value of an ecosystem service is evaluated in one or more regions, the benefits can be transferred also to other areas to obtain national or global level evaluations (benefit transfer). This is often the only option because of budget constraints.

The main difficulty in the benefit transfer is the question of how well the findings from one region can be generalized to the others (Table 1). For instance, the estimate of Costanza et al. (1997) on the total value of the global ecosystem services of 33 trillion US dollars annually has been widely criticized for the lack of attention to the context and consequently invalid benefit transfer (Bulte and van Kooten, 2000). However, such estimates potentially provide reasonable approximations and the main goal in the benefit transfer approach may not be accuracy, but usefulness (Richardson et al., 2015).

The transferability depends on the context and the spatial scale of the original studies (Richardson et al., 2015). For instance, it is possible to compute a unit value of one original evaluation study that best matches the characteristics of the site of interest and to use that value for the generalization. However, a meta-analysis modelling the ecosystem service values observed in the existing studies as a function of the characteristics of the study sites could be more useful. Such meta-analysis models are available for many ecosystem services (Nelson and Kennedy, 2009).

2.5. Expert judgment

Even if we seldom have suitable primary field or survey data for ecosystem services, it does not mean that we have no information at all. Instead, expert opinion may provide the best option available for assessing the ecosystem services in question for some cases.

Expert opinions are typically collected in the form of a model, meaning that the experts are asked to assess e.g. berry production potential, biodiversity, recreation value or habitat suitability in a given forest site type or forest age class, and the result is presented as a function of the forest characteristics. The most popular input data are probably the land use / land cover (LU/LC) matrices (Balzter et al., 2015). The expert opinions are typically assessed using qualitative techniques, such as on a verbal scale from very low to very high supply (Koschke et al., 2012, Fürst et al., 2013, Jacobs et al., 2015). It is also possible to use quantitative techniques such as pairwise comparisons, which enable producing statistical models from the judgments (Kangas et al., 1993, Ihalainen et al., 2002, Leskinen et al., 2003).

Information based on expert opinions is a less costly alternative compared to acquiring primary field or survey data on any ecosystem service. The drawback is that the quality of such data cannot be guaranteed. Large disagreements between the experts imply poor information quality (Kangas et al., 1998). The results depend on who, how many, and what kinds of experts are used. For instance, both scientists and tourists could be used as experts in questions concerning recreation. Techniques such as Delphi rounds have been used in order to increase the consensus between the experts (Scolozzi et al., 2012), but new rounds of assessments may also increase the variation between the experts (Kleemann et al., 2017). The judgments of experts could be weighted based on their expertise or the most uncertain experts could be left out (Leskinen et al., 2003). However, even when all experts agree on the provision of a given ecosystem service, they may be collectively wrong or the expert disagreeing with all the others could be the one with the best knowledge. This means that expert opinion is very susceptible to bias (Table 1).

2.6. Assessing the impact of changes

It is important to model the services based on information that allows for predicting their development in time and under human interventions – i.e., not just for mapping the current state. Land use change is the most important driver in the provision of ecosystem services and to predict consequences of land use changes, the provision of ecosystem services needs to be modelled as a function of the land use. In the simplest case, the experts assess the provision of ecosystem services directly as a function of LU/LC classes in two time points and the impacts of land use changes can be calculated as the subtraction of the before and after maps (e.g. Fürst et al., 2013, Kleeman et al., 2017). The landscape pattern also plays a major role in many cases, which means that the models need to account for the properties of the neighborhood of pixels or stands instead of just individuals.

A modelling approach can be used to capture the continuum of ecosystem services (Krishnaswamy et al., 2009). It is possible to model the provision of ecosystem services using either input variables fixed in time, such as topography, or those changing over time, such as canopy cover, tree species or biomass (e.g. Andrew et al., 2014, Martinez-Harms et al., 2016). Many types of modelling techniques, such as niche-based, trait-based or full process models based on actual causal relationships (Lavorel et al., 2017), have been used. The effect of land use change on the provision of ecosystem services can be predicted by first assessing the effect of the land use change on the input values. In a case of forest resources, matrix-based forest scenario models can be utilized (e.g. Vauhkonen and Packalen, 2017). Thus, it is possible to predict the consequences of different policies on ecosystem services by first predicting the future development of forest resources under these policies.

3. Mapping of forest ecosystems

Mapping ecosystem services means that the information available is generalized over an area based on remote sensing (RS) or other information presented in a Geographical Information System (GIS). The simplest – but often also the least accurate – way is to utilize previously collected and interpreted RS material, such as LU/LC maps provided by CORINE. It is possible to map all aspects of the cascade model (Fig. 1), in principle. In practice, however, parameters related to *in situ* production like carbon sequestration can be mapped fairly easily, whereas those related to benefits like climate regulation are used globally and thus mapping them is more challenging (Pagella and Sinclair, 2014).

As there is a lack of primary information concerning most of the ecosystem services, the most easily available map data source for ecosystem services is one based on expert assessment and LU/LC classes (Fig. 2, Fürst et al., 2010, 2013; Jacobs et al., 2015). The resulting map is inevitably heavily simplified. For instance, Metzger et al. (2006) considered all non-urban lands to have an equal potential for recreation and cropland and urban areas to have no recreational potential at all. Such maps also concentrate on the land use composition, but ignore land use configuration and intensity (Lavorel et al., 2017).

More detailed analyses require more detailed information, such as soil maps, vegetation or biotome maps or additional RS data (e.g. Vihervaara et al., 2015; Kaiser et al., 2013). For instance, Vauhkonen and Ruotsalainen (2017) used expert judgement based models to generalize the ecosystem service predictions derived from forest resource maps (e.g. Tomppo et al., 2008). Forest resource maps may often include more information than the LU/LC maps: beside the main tree species, also estimates of mean age, mean size and forest structure are included.

If primary data are available, the selected attributes can be generalized to larger areas by calculating the average estimates for the LU/LC classes instead of using expert assessments (e.g. Eigenbrod et al.,

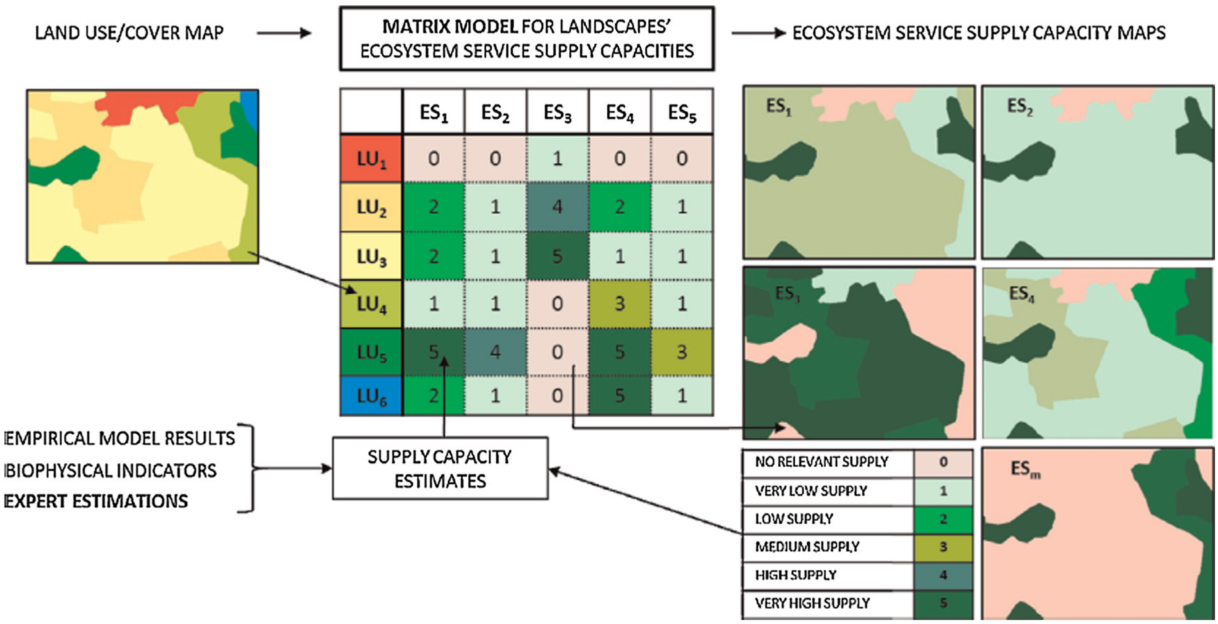

Fig. 2. Ecosystem service mapping based on LU/LC map and expert estimates of ecosystem services provision in the classes (Jacobs et al., 2015).

2010). The use of average within-class estimates may over-simplify the situation, as forests belonging to the same LU/LC class may have a very different potential to provide ecosystem services according to stand mean age, for example. Primary data can also be used in physical or statistical models, where the provision of ecosystem services is predicted for each pixel or stand using available GIS and RS information (e.g. Andrew et al., 2014; Martinez-Harms et al., 2016). Such models can be directly utilized in mapping.

The accuracy of the produced map improves by a stronger relationship between the input variables and the ecosystem services in question. Thus, three-dimensional (3D) remote sensing products such as airborne laser scanning (ALS) data allow producing most detailed maps of forest ecosystem services (Davies and Asner, 2014; Vihervaara et al., 2015). Unfortunately, the availability of 3D remote sensing data seldom supports large-scale analyses in the same way as the availability of satellite images.

4. Uncertainties in the ecosystem services mapping

4.1. Assessing the uncertainty

There may be uncertainties due to indicators chosen to describe the ecosystem services; GIS data used for mapping the services; relationships between the GIS data and the indicators; and due to the relationships between human interventions and the ecosystem services (Table 2). Unfortunately, the accuracy assessment protocols used with map products do not clearly inform on if a map is useful or not (Ayanu et al., 2012; McRoberts, 2011; Pagella and Sinclair, 2014). The first and last categories of Table 2 apply to decision making involving ecosystem services overall, and the other two categories for decisions concerning location-specific decision making.

The uncertainties can be random or non-random (or structural, see Boithias et al., 2016). Non-random errors can be due to e.g. incoherent definitions of the ecosystem services, selection of indicators, spatial scale, missing explanatory variables, or analysis techniques (Martínez-Harms and Balvanera, 2012; Schulp et al., 2014; Barredo et al., 2015). Non-random errors are also inevitable whenever expert judgment is used to assess the ecosystem service, e.g. due to the selection of the experts. In the valuation of ecosystem services, the number of services considered and the number of benefits evaluated for each service may introduce non-random errors (Boithias et al., 2016). Random errors, on the other hand, come from measurement errors, sampling errors and residual errors of models.

The first source of uncertainty in the ecosystem services estimates is the use of primary data (Table 3). In the case of random errors, basic statistical analyses of measurement and sampling errors are valid assessment methods applied for indicators such as timber and carbon production. Non-random errors are much more difficult to assess, but empirical validation studies could be used to assess the effect of selected indicators or the variations of indicators between regions. Such studies could also be used to analyze the effects of missing indicators. The vagueness of definitions could be described using e.g. fuzzy numbers or sets, but from decision making point of view the best approach would be to refine the definitions. On the other hand, the variation of the measures used to describe the ES in question could in itself describe the vagueness of the definitions (Boerema et al., 2017). Uncertainty in the relationship of the chosen indicator and the ecosystem service in question is more complicated. To be useful, the indicator should have a high positive correlation with the service in question. With empirical validation data, it is possible to analyze the degree by which the chosen indicators succeed.

The second source of uncertainty is the actual GIS data used for mapping, which can contain random errors due to misclassifications of the LU/LC classes, for instance. Such uncertainty can be assessed using Monte Carlo simulation (e.g. Dong et al., 2015; Foody, 2015). The random errors can also be prediction errors, resulting from model-based generalizations of e.g. soil samples across regions. The non-random errors of GIS data may be related to varying amounts of LU/LC classes or varying resolution in the RS material between different studies. Assessing such uncertainties in the context of ecosystem mapping has largely been carried out by comparing estimates of ecosystem services obtained from different land use maps (e.g. Benítez et al., 2007; Schulp and Alkemade, 2011; Schulp et al., 2014; Van der Biest et al., 2015). For instance, Schulp et al. (2014) compared four maps at the European level. As a result of the uncertainties, the produced maps strongly disagreed on the potential of climate regulation in Sweden and Finland despite the high proportion of forests in these countries. In such a comparison, the problem is that none of the maps is necessarily the true one and the accuracy of each map remains unknown. Another problem is that the varying sources of error are confounded and it is typically not possible to separate the effects of just one source.

The third source of error is the relationship between the indicators

Table 2
Sources of errors in the mapping.

Source of error	Random error	Non-random errors
Indicators of ecosystem services	• Measurement errors of indicator variables • Sampling errors • Poor correlation between the indicator and the ES	• Ambiguous definitions of ES • Selection of indicators • Variation in indicators used for a given ES across regions and studies • Misspecified relationship between the indicator and the ES
GIS data (used as a basis for mapping)	• Misclassification errors	• Number of LU/LC classes • Spatial resolution of the data • Missing data
Relationship between GIS data and indicators of ES	• Residual errors of models (poor correlation) • Errors in model coefficients	• Selection of experts • Bias in expert judgment • Misspecified (statistical or expert) models • Missing explanatory variables • Regional bias (in benefit transfer and other models)
Impact of human interventions on ES	• Residual errors of models (poor correlation) • Errors in model coefficients	• Selection of experts • Bias in expert judgment • Misspecified (statistical or expert) models • Missing explanatory variables • Regional bias

Table 3
Methods suitable for assessing different types of errors.

Source of error	Assessment of random errors	Assessment of non-random errors
Indicators of ecosystem services	• Statistical analysis of sampling and measurement errors • Empirical validation of correlation between the service and the indicator	• Fuzzy logic • Empirical validation of selection of indicators used
GIS data (used as a basis for mapping)	• Statistical analysis of model errors • Monte Carlo simulation	• Comparison of maps produced
Relationship between GIS data and indicators of ESImpact of human interventions	• Statistical analysis of model errors • Monte Carlo simulation • Empirical validation of correlation between the service and the indicator	• Variation among experts • Variation among prediction models • Empirical validation of model misspecification

of ecosystem services and the GIS data used for the mapping. The random errors are due to the prediction models between input data and the primary data on indicators of the ecosystem services. Non-random errors are due to, for instance, using expert judgment for modelling. Assessing the uncertainties in expert assessments of ecosystem services is not self-evident, but one way to assess the uncertainties is to analyze the disagreements between the experts using statistical methods (Alho and Kangas, 1997, Table 3). Even if primary data are available and the relationship is modelled with statistical models, non-random errors due to missing explanatory variables or other model misspecifications is possible, even likely. That source of error can be assessed by comparing different models in the predictions.

The errors in the relationships between variables can be assessed using empirical evaluation. Land cover-based proxies are known to be poor predictors, because they have a poor correlation with the actual ecosystem services (Eigenbrod et al., 2010; Geijzendorffer and Roche, 2013). For instance, Eigenbrod et al. (2010) generalized sample-based data of three ecosystem services to 10×10 km squares and compared the resulting maps to those produced as averages for the land cover classes. Both the approaches were thus based on the same data, but classified either with or without using the land cover classes. When the areas with highest potential for provision of ecosystem services were searched for, the congruence between these two maps was not very high. For instance, when looking at hotspots (the best 10% of the area), the two maps were overlapping in 23% of cases for biodiversity, 17% for recreation and 62% for carbon storage.

A model that uses local biophysical information as explanatory variables in addition to land use can produce much more accurate information. For instance, in the comparison carried out by Martinez-Harms et al. (2016), a model could predict 60% of the variation of the firewood, compared to the 15% of the look-up-table. With this kind of model, it is also possible to analyze the importance of the different components of uncertainty (e.g. Livne and Svoray, 2011).

The last source of error is the relationship between the ecosystem service production and human interventions. The assessment methods for this source of error are similar as in the previous case.

The uncertainty assessments should be spatially explicit instead of aggregated in applications aiming to use the ecosystem maps to locate the areas of hotspots. One problem is that the errors may be very heterogeneously distributed over the landscape or between the different ecosystem services considered (Dong et al., 2015). For instance, Grêt-Regamey et al. (2013) used a Bayesian network for a spatially explicit uncertainty assessment. It is also possible to map the robustness and sensitivity of the expert opinions (Ligmann-Zielinska and Jankowski, 2014; García Márquez et al., 2017).

The uncertainties from the different sources cumulate in the analysis (e.g. Barton et al., 2018). For instance, if ecosystem structure is used as a basis for assessing ecosystem function(s), which is further used as a basis for predicting the benefit(s) and value, the errors related to the different steps of the cascade model cumulate. This cumulating effect thus mainly affects the primary data concerning the benefit(s) and value. However, the cumulating effect also applies to cases where decisions are carried out based on locally predicted impacts of human interventions, which in turn are based on the predicted provision of the ecosystem services. In such cases, all sources of errors introduced in Table 2 apply, but the users of the information should especially be cautioned on the cumulating effect of the non-random errors.

4.2. The effect of uncertainty in decision making

The uncertainties should be quantified and communicated to decision makers in order to make informed decisions. Errors can lead to biased valuation of the ecosystem services and, consequently, erroneous decisions (Foody, 2015). The distinction between random and non-

Table 4
Possibilities to account for the errors in decision making.

Source of error	Random errors in decision analysis	Non-random errors in decision analysis
Indicators of ecosystem services	• Sensitivity analysis • Scenario analysis • Stochastic optimization • Bayesian decision analysis	• Fuzzy decision analysis • Fuzzy optimization
GIS data (used as a basis for mapping)	• Sensitivity analysis • Scenario analysis • Stochastic optimization • Bayesian decision analysis	• Scenario analysis
Relationship between GIS data and indicators of ES	• Sensitivity analysis • Scenario analysis • Stochastic optimization • Bayesian decision analysis	• Scenario analysis
Impact of human interventions	• Sensitivity analysis • Scenario analysis • Stochastic optimization • Bayesian decision analysis	• Adaptive management • Scenario analysis

random errors (Table 4) is especially important when accounting for the uncertainties. For all random error types, it is possible to simulate several outcomes of a given decision and use these simulations either for sensitivity or scenario analyses. When scenarios can be simulated, it is also possible to use stochastic programming or Bayesian decision analysis for the decision making. Stochastic programming makes sense when there are constraints in the analysis (e.g. Eyvindson and Kangas, 2014; Hartikainen et al., 2016). Bayesian analysis is especially useful if the problem can be described using a decision tree with only a few possible outcomes for each decision (Smith, 2012).

Scenario analyses with scenarios based on the assessments of different experts or different models are also one possibility to assess the non-random errors, such as uncertainties in the expert judgments. In the assessments of the effects of human interventions, it is possible to utilize adaptive management, i.e. learn from experiences and adapt the management accordingly (Birge et al., 2016). However, adaptive management based on two-stage stochastic programming is also a possibility, when additional information is acquired (Kangas et al., 2015).

Vague definitions are the most challenging non-random error, but decision support methods such as fuzzy optimization and fuzzy decision analysis (Kangas et al., 2015) can still be utilized when all uncertainties are described using fuzzy approach rather than statistical analysis.

Different types of uncertainties may be important for the different types of decisions. Therefore, it may not be necessary to account for all types of errors in every decision problem. As the effects of these errors are very poorly known, it is difficult to assess the importance of each error source on the decisions. However, based on the map typologies of Pagella and Sinclair (2014), we have assessed the importance of accounting for errors for some applications. Table 5 presents the results of this analysis using three classes (low, medium, or high), which indicate how important it is to include uncertainty analyses in the example applications for those to produce truthful outcomes.

Random errors in primary data for the indicators and GIS data likely have a small effect on trade-off analyses between different ecosystem services. In contrary, the most important source of error for the trade-off analyses is the relationship between GIS data and the indicators of the ecosystem service. The non-random errors are especially important. If the relationship is biased, it is evident that the relationship between two ecosystem services analyzed for trade-offs will be biased as well (Table 5). Likewise, vague definitions of ecosystem services in the first place are also likely very important. However, the best way to avoid this source of uncertainty is using unambiguous definitions.

On the other hand, the accuracy of the locations is important, if we wish to find hot or cold spots for the considered ecosystem services. In such case, also the random errors in the GIS data may be important. However, the non-random errors in the relationships between the GIS data and the indicators of ecosystem services are likely to be the most important sources of error when detecting the hot spots. Obviously, the relationships between the impacts and the ecosystem services are more important than in the previous decision types, if we wish to map the impacts. The uncertainties related to the impacts are also important, if we wish to detect locations with greatest possibilities for increasing the provision of selected ecosystem services. The uncertainties in the GIS data are also fairly important in detecting the correct spots. Here, as well as in the previous cases, we assess the non-random uncertainties to have the highest impacts in the analyses.

5. Discussion

Considering the high relevancy of ecosystem services for policy making, the information basis is not adequate for most of the forest-related ecosystem services. Maps of ecosystem services are only as good as the data available on the services to be included (Blackstock et al., 2015). Therefore, it is concerning that in a recent review of 405 papers on ecosystem services (Boerema et al., 2017), only a fifth of the studies included actual field measurements, i.e. primary data, and 31% of studies did not report any data.

It is possible to make informed decisions concerning some of the forestry-related services, especially timber production and carbon sequestration for which good-quality data are usually available from NFIs. For some others, like berry and mushroom production, information is available, but its quality may not be high enough for informed decisions (see Kilpeläinen et al., 2016). Maps concerning the benefits and values also involve considerable errors (Schägner et al., 2013). Moreover, the quality of the indicators themselves should be validated. Many of the indicators currently used are very weakly correlated with the ecosystem services (Boerema et al., 2017). Measuring indicator variables in national and global forest inventories would provide a breakthrough for sustainable forest management (e.g. Miura et al., 2015). It would mean that primary data on the services are available both for the analyses and for the validation.

Assessing the uncertainties in the estimated provision of the ecosystem services is considered to be highly important (e.g. Boerema et al., 2017, Barton et al., 2018). In spite of this, Boerema et al. (2017) conclude that uncertainty and validation analyses are mostly ignored. Even if uncertainties are assessed, the assessment typically covers only one or two possible sources of uncertainty. Many other potential sources listed in the previous sections have only rarely been comprehensively addressed. In the papers we reviewed, the non-random uncertainties related to the number of LU/LC classes and RS data resolution were addressed most often. We also discovered cases where the random misclassification errors of GIS data, the non-random uncertainty due to possible model misspecification, and the random uncertainty due to poor correlation were addressed. Such cases were still rare, considering that our search was specifically focused on papers addressing the uncertainty. It is probable that we did not find all relevant papers, but considering the obvious lack of existence of such papers, unlikely many were missed.

For some ecosystem services, the only available analyses are based on expert opinion and LU/LC classes, which may be highly biased and poorly correlated with the actual services. Considering that primary data are available only for a minority of studies, the expert opinion is a very important source of data. Related to the importance of this data source, it is notable that we were not able to find any ES studies that

Table 5
Assessment of the importance of uncertainty assessment for various example applications (R – random and N – non-random errors).

Application	Source of error							
	Indicators of ecosystem services		GIS data (used as a basis for mapping)		Relationship between GIS data and ES		Impact of human interventions	
	R	N	R	N	R	N	R	N
Analyses of trade-offs and synergies	Low	High	Low	Low	Medium	High	Low	Low
Detecting cold and hot spots	Low	High	Medium	Medium	Medium	High	Low	Low
Mapping of impacts	Low	High	Low	Low	Medium	High	Medium	High
Mapping of opportunities for improvements	Low	High	Medium	Medium	Medium	High	Medium	High

would have addressed the uncertainty due to (non-random) variation among the experts or possible biases or misspecifications of the expert opinion models. In the future, it is very important to validate these expert judgment studies. As the majority of experts can be wrong, it is important to exercise caution with these analyses.

There is an inevitable tradeoff between the coverage, level of detail, and accuracy of the data, which reflects to the decision making. For international policy making, global assessments are required and a low level of detail and low accuracy might be acceptable. The requirements of detail and accuracy increase from the national to regional and local levels of policy making. Uncertain ecosystem service maps with a low level of detail are ill-suited for decision support or for identifying areas that produce multiple services at a local level. Even so, such maps might still be used to detect large-scale trends (Eigenbrod et al., 2010). When local primary data are available for the mapping, maps that are better suited also for local decision making can be obtained.

The uncertainties in the ecosystem services analyses are high, but producing significantly more accurate information on ecosystem services may take years. Nevertheless, Costanza et al. (2017) proposed improving the measurements and their utilization as a key to identify differences in outcomes among policy choices. Moore et al. (2017) recommend that in some cases qualitative rather than quantitative information on ecosystem services is used, due to the high uncertainty. We agree with the problem brought up by both the references above, but disagree on means to solve it. Already in the short term and in the absence of better measurements, appropriate and transparent information can be provided to decision makers using quantitative information and decision analysis techniques that account for the inherent uncertainties (Table 4).

The quality requirements concerning the information on ecosystem services for planning and policy making should be assessed. This assessment depends on the scale of the decision problem (local, regional or national). It is also important to relate the quality needs to the temporal scale of the decisions and policies. It is clear that in a spatial scale of pixels, stands or estates, the decisions to be made are different than those for national or global levels. As a consequence, also the indicators that are relevant and/or useful are different, as well as the accuracy requirements set to these indicators. We also need to assess the quality requirements separately for different types of problems (Table 5), as the data requirements are obviously different for different tasks. The most pressing need is to assess non-random uncertainties.

The sufficient accuracy of information can be addressed based on the concept of value of information (VOI) for each specific decision problem. It can (ex ante) be calculated as the difference of expected value of a given decision with and without a source of new information (Lawrence, 1999, Kangas, 2010). The VOI for monitoring ecosystem services has not been estimated in any study to our knowledge. However, Eigenbrod et al. (2010) conclude that benefits from improving the information on ecosystem services by sample-based mapping far outweight the costs.

There is also an urgent need to communicate the scope and limitations of various ecosystem service maps to the users (Vorstius and Spray, 2015). Moreover, there is a need to introduce the uncertainties into the decision making process, in order to improve the actual decisions. Unfortunately, Barton et al. (2018) conclude that only 2% of the 313 papers they reviewed actually targeted decision making. Addressing the uncertainty in the decision process could improve the decisions and reduce the risks in a more cost-efficient way than improving the accuracy of predictions by better primary data (e.g. Eyvindson and Kangas, 2014). This is possible when the decisions are good for many scenarios of the future provision of ecosystem services, instead of being the best or optimal for just one outcome. There is a need to change the focus from assessing and mapping the ecosystem services to actually using the information in decision making, and doing it intelligently, i.e. taking also the uncertainties into account.

6. Conclusions

The quality of information for different ecosystem services varies between the different services. Forest inventories provide accurate information for timber production, whereas for many other services, expert opinions serve as the main source of information. The larger the spatial scale of the analyses, the less information is typically available and this situation cannot be expected to markedly improve in the future. Incorporating the uncertainties into the decision making and policy analyses enables us to produce robust decisions and policies, meaning that the recommendations apply to a whole set of possible scenarios of future and very poor outcomes can be avoided. Thus, assessing the uncertainties and acknowledging them in decision making is the fastest and cheapest way to improve the provision policies of forest-related ecosystem services.

References

Alho, J.M., Kangas, J., 1997. Analyzing uncertainties in experts' opinions of forest plan performance. Forest Sci. 43, 521–528.
Andrew, M.E., Wulder, M.A., Nelson, T.A., 2014. Potential contributions of remote sensing to ecosystem service assessments. Progr. Phys. Geogr. 38, 328–353.
Auvinen, A.-P., Hildén, M., Toivonen, H., Primmer, E., Niemelä, J., Aapala, K., Bäck, S., Härmä, P., Ikävalko, J., Järvenpää, E., Kaipiainen, H., Korhonen, K.T., Kumela, H., Kärkkäinen, L., Lankoski, J., Laukkanen, M., Mannerkoski, I., Nuutinen, T., Nöjd, A., Punttila, P., Salminen, O., Söderman, G., Törmä, M., Virkkala, R., 2007. Evaluation of the Finnish National Biodiversity Action Plan 1997-2005. Monographs of the Boreal Environment Research No. 29. 54 p.
Ayanu, Y.Z., Conrad, C., Nauss, T., Wegmann, M., Koellner, T., 2012. Quantifying and mapping ecosystem services supplies and demands: a review of remote sensing applications. Environ. Sci. 46, 8529–8541.
Balzter, H., Cole, B., Thiel, C., Schmullius, C., 2015. Mapping CORINE land cover from sentinel-1A SAR and SRTM digital elevation model data using random forests. Remote Sens. 7, 14876–14898.
Barredo, J.I., et al., 2015. Mapping and assessment of forest ecosystems and their services – Applications and guidance for decision making in the framework of MAES. Report EUR 27751 EN, Joint Research Centre, European Union, 78 p. doi: 10.2788/720519.
Barton, D.N., Kelemen, E., Dick, J., Martin-Lopez, E., Gómez-Baggethun, S., Jacobs, S., Hendricks, C.M.A., Termansen, M., García-Llorente, M., Primmer, E., Dunford, R., Harrison, P.A., Turkelboom, F., Saarikoski, H., van Dijk, J., Rusch, G.M., Palomo, I., Yli-Pelkonen, V.J., ... Lapola, D.M., 2018. (Dis) integrated valuation – Assessing the information gaps in ecosystem service appraisals for governance support. Ecosyst. Serv. 29 (C), 529–541.

Benítez, B.C., McCallum, I., Obersteiner, M., Yamagata, Y., 2007. Global potential for carbon sequestration: Geographical distribution, country risk and policy implications. Ecol. Econ. 60, 572–583.
Birge, H.E., Allen, C.R., Garmestani, A.S., Pope, K.L., 2016. Adaptive management of ecosystem services. J. Environ. Manage. 183, 343–352.
Blackstock, K., Martin-Ortega, J., Spray, C.J., 2015. Implementation of the European water framework directive: what does taking an ecosystem services-based approach add? In: Martin-Ortega, J., Ferrier, R.C., Gordon, I.J., Khan, S. (Eds.), Water Ecosystem Services: A Global Perspective. Cambridge University Press. (International Hydrology Series), Cambridge, pp. 57–64.
Boerema, A., Rebelo, A.J., Bodi, M.B., Esler, K.J., Meire, P., 2017. Are ecosystem services adequately quantified? J. Appl. Ecol. 54, 358–370.
Boithias, L., Terrado, M., Corominas, L., Ziv, G., Kumar, V., Margués, M., Schumacher, M., Acuña, V., 2016. Analysis of the uncertainty in the monetary valuation of ecosystem services – A case study at the river basin scale. Sci. Total Environ. 543, 683–690.
Bulte, E., Van Kooten, G.C., 2000. Economic science, endangered species, and biodiversity loss. Conserv. Biol. 14, 113–119.
Chirici, G., McRoberts, R.E., Winter, S., Bertini, R., Braendli, U.-B., Asensio, I.A., Bastrup-Birk, A., Rondeux, J., Barsoum, N., Marchetti, M., 2012. National forest inventory contributions to forest biodiversity monitoring. Forest Sci. 58, 257–268. http://dx.doi.org/10.5849/forsci.12-003.
Chirici, G., Winter, S., McRoberts, R.E. (Eds.), 2011. National Forest Inventories: Contributions to Forest Biodiversity Assessments. Springer. Managing Forest Ecosystems.
Costanza, R., d'Arge, R., de Groot, R., Farberk, S., Grasso, M., Hannon, B., Limburg, K., Naeem, S., O'Neill, R.V., Paruelo, J., Raskin, R.G., Sutton, P., van den Belt, M., 1997. The value of the world's ecosystem services and natural capital. Nature 387 (6630), 253–260.
Costanza, R., de Groot, R., Braat, L., Kubiszewski, I., Fioramonti, L., Sutton, P., Farber, S., Grasso, M., 2017. Twenty years of ecosystem services: how far have we come and how far do we still need to go? Ecosyst. Serv. 28, 1–16.
Costanza, R., 2008. Ecosystem services: Multiple classification systems are needed. Biol. Conserv. 141, 350–352.
Corona, P., 2016. Consolidating new paradigms in large-scale monitoring and assessment of forest ecosystems. Env. Res. 144, 8–14. http://dx.doi.org/10.1016/j.envres.2015.10.017.
Corona, P., Chirici, G., McRoberts, R.E., Winter, S., Barbati, A., 2011. Contribution of large-scale forest inventories to biodiversity assessment and monitoring. For. Ecol. Manage. 262, 2061–2069. http://dx.doi.org/10.1016/j.foreco.2011.08.044.
Crossman, N.D., Burkhard, B., Nedkov, S., Willemen, L., Petz, K., Palomo, I., Drakou, E.D., Martín-Lopez, B., McPhearson, T., Boyanova, K., Alkemade, R., Egoh, B., Dunbar, M.B., Maes, J., 2013. A blueprint for mapping and modelling ecosystem services. Ecosyst. Serv. 4, 4–14.
Daily, G. (Ed.), 1997. Nature's Services: Societal Dependence on Natural Ecosystems. Island Press.
Davies, A.B., Asner, G.P., 2014. Advances in animal ecology from 3D-LiDAR ecosystem mapping. Trends Ecol. Evol. 29, 681–691.
Dong, M., Bryan, B.A., Connor, J.D., Nolan, M., Gao, L., 2015. Land use mapping error introduces strongly-localized, scale-dependent uncertainty into land use and ecosystem services modelling. Ecosyst. Serv. 15, 63–74.
Eigenbrod, F., Armsworth, P.R., Anderson, B.J., Heinemeyer, A., Gillings, S., Roy, D.B., Thomas, C.D., Gaston, K.J., 2010. The impact of proxy-based methods on mapping the distribution of ecosystem services. J. Appl. Ecol. 47, 377–385.
European Commission, 2011. Communication from the Commission to the European Parliament, the Council. The Economic and Social Committee and the Committee of the Regions, Brussels, 3.5.2011 COM(2011) 244 final.
Eyvindson, K., Kangas, A., 2014. Stochastic goal programming in forest planning. Can. J. For. Res. 44, 1274–1280.
FAO, CIFOR, IFRI and World Bank. 2016. National socioeconomic surveys in forestry: guidance and survey modules for measuring the multiple roles of forests in household welfare and livelihoods, by R.K. Bakkegaard, A. Agrawal, I. Animon, N. Hogarth, D. Miller, L. Persha, E. Rametsteiner, S. Wunder, A. Zezza. FAO Forestry Paper No. 179. Food and Agriculture Organization of the United Nations, Center for International Forestry Research, International Forestry Resources and Institutions Research Network and World Bank.
Foody, G.M., 2015. Valuing map validation: The need for rigorous land cover map accuracy assessment in economic valuations of ecosystem services. Ecol. Econ. 111, 23–28.
Frank, S., Fürst, C., Koschke, L., Makeschin, F., 2012. A contribution towards a transfer of the ecosystem service concept to landscape planning using landscape metrics. Ecol. Ind. 21, 30–38.
Frank, S., Fürst, C., Pietzsch, F., 2015. Cross-sectoral resource management: how forest management alternatives affect the provision of biomass and other ecosystem services. Forests 6, 533–560. http://dx.doi.org/10.3390/f6030533.
Fürst, C., Volk, M., Pietzsch, K., Makeschin, F., 2010. Pimp your landscape: a tool for qualitative evaluation of the effects of regional planning measures on ecosystem services. Environ. Manage. 46, 953–968.
Fürst, C., Frank, S., Witt, A., Koschke, L., Makeschin, F., 2013. Assessment of the effects of forest land use strategies on the provision of ecosystem services at regional scale. J. Environ. Manage. 127.
Fürst, C., 2015. Does using the ecosystem services concept provoke the risk of assigning virtual process instead of real values to nature? Some reflections on the benefit of ecosystem services for planning and policy consulting. Eur. J. Ecol. 1, 39–44.
García Márquez, J.R., Krueger, T., Páez, C.A., Ruiz-Agudelo, C.A., Bejarano, P., Muto, T., Arjona, F., 2017. Effectiveness of conservation areas for protecting biodiversity and ecosystem services: a multi-criteria approach. Int. J. Biodiversity Sci., Ecosyst. Services Manage. 13 (1), 1–13.
Geijzendorffer, I.R., Roche, P.K., 2013. Can biodiversity monitoring schemes provide indicators for ecosystem services? Ecol. Indicators 33, 148–157.
Grêt-Regamey, A., Brunner, S.H., Altwegg, J.H., Bebi, P., 2013. Facing uncertainty in ecosystem services-based resource management. Ecosyst. Serv. 15, 63–74.
de Groot, R.S., Wilson, M.A., Boumans, R.M.J., 2002. A typology for the classification, description and valuation of ecosystem functions, goods and services. Ecol. Econ. 41, 393–408.
Haakana, H., Hirvelä, H., Hanski, I.K., Packalen, T., 2017. Comparing regional forest policy scenarios in terms of predicted suitable habitats for the Siberian flying squirrel (Pteromys volans). Scand. J. For. Res. 32, 185–195.
Haines-Young, R., Potschin, M., 2010. Proposal for a Common International Classi-fication of Ecosystem Goods and Services (CICES) for Integrated Environmentaland Economic Accounting. Background Document, Report to the EEA (21 March2010).
Hansen, K., Malmaeus, M., 2016. Ecosystem services in Swedish forests. Scand. J. For. Res. 31, 626–640.
Japelj, A., Mavsar, R., Hodges, D.G., Kovač, M., Juvančič, L., 2016. Latent preferences of residents regarding an urban forest recreation setting in Ljubljana, Slovenia. Forest Policy Econ. 71, 70–78.
Hartikainen, M., Eyvindson, K., Miettinen, K., Kangas, A., 2016. Data-based forest management with uncertainties and multiple objectives. In: Giuffrida, G., Nicosia, G., Pardalos, P. (Eds.), The Second International Workshop on Machine Learning, Optimization and big Data - MOD 2016. Springer LNCS 10122.
Horne, P., Boxall, P.C., Adamowicz, W.L., 2005. Multiple-use management of forest recreation sites: a spatially explicit choice experiment. Forest Ecol. Manage. 207, 189–199.
Ihalainen, M., Alho, J., Kolehmainen, O., Pukkala, T., 2002. Expert models for bilberry and cowberry yields in Finnish forests. For. Ecol. Manage. 157, 15–22.
Jacobs, S., Burkhard, B., Van Daele, T., Staes, J., Schneiders, A., 2015. 'The Matrix Reloaded': A review of expert knowledge use for mapping ecosystem services. Ecol. Modellling 295, 21–30.
Kaiser, G., Burkhard, B., Römer, H., Sangkaew, S., Graterol, R., Haitook, T., Sterr, H., Sakuna-Schwartz, D., 2013. Mapping tsunami impacts on land cover and related ecosystem service supply in Phang Nga, Thailand. Nat. Hazards Earth Syst. Sci. 13, 3095–3111.
Kangas, A., Kurttila, M., Hujala, T., Eyvindson, K., Kangas, J., 2015. Decision support for forest management. In: Managing Forest Ecosystems, second ed. Springer, pp. 307.
Kangas, A., 2010. Value of forest information. Eur. J. Forest Res. 129, 863–874.
Kangas, J., Alho, J., Kolehmainen, O., Mononen, A., 1998. Analyzing consistency of experts' judgments - Case of assessing forest biodiversity. Forest Sci. 44 (4), 610–617.
Kangas, J., Laasonen, L., Pukkala, T., 1993. A method for estimating forest landowner's landscape preferences. Scand. J. For. Res. 8, 408–417.
Kant, S., Lee, S., 2004. A social choice approach to sustainable forest management: an analysis of multiple forest values in Northwestern Ontario. Forest Policy Econ. 6, 215–227.
Kilpeläinen, H., Miina, J., Store, R., Salo, K., Kurttila, M., 2016. Evaluation of bilberry and cowberry yield models by comparing model predictions with field measurements from North Karelia, Finland. For. Ecol. Manage. 363, 120–129.
Kleeman, J., Baysal., G., Bulley, H.N.N. and Fürst, C., 2017. Assessing driving forces of land use and land cover changes by a mixed-method approach in northern-eastern Ghana, West Africa. J. Environ. Manage. 196, 411–442.
Koschke, L., Fürst, C., Frank, S., Makechin, F., 2012. A multi-criteria approach for an integrated land-cover-based assessment of ecosystem services provision to support landscape planning. Ecol. Ind. 21, 54–66.
Kurki, S., Nikula, A., Helle, P., Lindén, H., 2000. Effects of landscape fragmentation and forest composition on breeding success of grouse in boreal forests. Ecology 81, 1985–1997.
Krishnaswamy, J., Bawa, K.S., Ganeshaiah, K.N., Kiran, M.C., 2009. Quantifying and mapping biodiversity and ecosystem services: Utility of a multi-season NDVI based Mahalanobis distance surrogate. Remote Sens. Environ. 113, 857–867.
Lavorel, S., Bayer, A., Bondeau, A., Lautenbach, S., Ruiz-Frau, A., Schulp, N., Seppelt, R., Verburg, P., van Teeffelen, A., Vannier, C., Arneth, A., Cramer, W., Marba, N., 2017. Pathways to bridge the biophysical realism gap in ecosystem services mapping approaches. Ecol. Ind. 74, 241–260.
Lawrence, D.B., 1999. The Economic Value of Information. Springer, pp. 393.
Leskinen, P., Kangas, J., Pasanen, A.-M., 2003. Assessing ecological values with dependent explanatory variables in multi-criteria forest ecosystem management. Ecol. Model. 170, 1–12.
Ligmann-Zielinska, A., Jankowski, P., 2014. Spatially-explicit integrated uncertainty and sensitivity analysis of criteria weights in multicriteria land suitability evaluation. Environ. Modell. Software 57, 235–247.
Livne, E., Svoray, T., 2011. Components of uncertainty in primary production model: the study of DEM, classification and location error. Int. J. Geographical Inf. Sci. 25, 473–488.
Louviere, J.D., Hensher, D., Swait, J., 2000. Stated Choice Methods. Analysis and Application. Cambridge University Press, pp. 402.
M.A., 2005. Millennium ecosystem assessment. In: Ecosystems and Human Well-Being: Synthesis. Island Press, Washington, D.C.
Martinez-Harms, M.J., Quijas, S., Merenlender, A.M., Balvanera, P., 2016. Enhancing ecosystem services maps combining field and environmental data. Ecosyst. Serv. 22, 12–40.
Martinez-Harms, M.J., Bryan, B.A., Balvanera, P., Law, E.A., Rhodes, J.R., Possingham, H.P., Wilson, K.A., 2015. Making decisions for managing ecosystem services. Biol. Conserv. 184, 229–238.
Martínez-Harms, M.J., Balvanera, P., 2012. Methods for mapping ecosystem service supply: a review. Int. J. Biodiversity Sci., Ecosyst. Serv. Manage. 8 (1–2), 17–25.

Maes, J., Paracchini, M.L., Zulian, G., Dunbar, M.B., Alkemade, R., 2012a. Synergies and trade-offs between ecosystem service supply, biodiversity and habitat conservation status in Europe. Biol. Conserv. 155, 1–12.
Maes, J., Egoh, B., Willemen, L., Liquete, C., Vihervaara, P., Schägner, J.P., Grizzetti, B., Drakou, E.G., LaNotte, A., Zulian, G., Bouraoui, F., Paracchini, M.L., Braat, L., Bidoglio, G., 2012b. Mapping ecosystem services for policy support and decision making in the European Union. Ecosyst. Serv. 1, 31–39.
McRoberts, R.E., 2011. Satellite image-based maps: Scientific inference or pretty pictures? Remote Sens. Environ. 115, 715–724.
Melin, M., Mehtätalo, L., Miettinen, J., Tossavainen, S., Packalen, P., 2016. Forest structure as a determinant of grouse brood occurrence - An analysis linking LiDAR data with presence/absence field data. For. Ecol. Manage. 380, 202–211.
Menzel, S., Teng, J., 2010. Ecosystem services as a stakeholder-driven concept for conservation science. Conserv. Biol. 24, 907–909.
Metzger, M.J., Rounsevell, M.D.A., Acosta-Michlik, L., Leemans, R., Schrotere, D., 2006. The vulnerability of ecosystem services to land use change. Agric. Ecosyst. Environ. 114, 69–85.
Meyer, M.A., Schultz, C., 2017. Do ecosystem services provide an added value compared to existing forest planning approaches in Central Europe? Ecosyst. Soc. 22, 6.
Miura, S., Amacher, M. San-Miguel-Ayanz, J., Ernawati, Thackway R., 2015. Protective functions and ecosystem services of global forests in the past quarter-century. For. Ecol. Manage. 352, 35–46.
Mononen, L., Auvinen, A.-P., Ahokumpu, A.-L., Rönkä, M., Aarras, N., Tolvanen, H., Kamppinen, M., Viirret, E., Kumpula, T., Vihervaara, P., 2016. National ecosystem service indicators: Measures of social–ecological sustainability. Ecol. Ind. 61, 27–37.
Moore, D.W., Booth, P., Alix, A., Apitz, S.E., Forrow, D., Huber-Sannwald, E., Jayasundara, N., 2017. Application of ecosystem services in natural resource management decision making. Integr. Environ. Assess. Manage. 13, 74–84.
Müller, F., Burkhart, B., 2012. The indicator side of ecosystem services. Ecosyst. Serv. 1, 26–30.
Nelson, J.P., Kennedy, P.E., 2009. The use (and abuse) of meta-analysis in environmental and natural resource economics: an assessment. Environ. Resour. Econ. 42 (3), 345–377.
Norgaard, R.B., 2010. Ecosystem services: From eye-opening metaphor to complexity blinder. Ecol. Econ. 69, 1219–1227.
Pagella, T.F., Sinclair, F.I., 2014. Development and use of typology of mapping tools to assess their fitness for supporting management of ecosystem service provision. Landscape Ecol. 29, 383–399.
Pukkala, T., Nuutinen, T., Kangas, J., 1995. Integrating scenic and recreational amenities into numerical forest planning. Landscape Urban Plann. 32, 185–195.
Ramezani, H., Ramezani, F., 2015. Potential for the wider application of national forest inventories to estimate the contagion metric for landscapes. Environ. Monit. Assess. 187, 116. http://dx.doi.org/10.1007/s10661-015-4283-2.
Richardson, L., Loomis, J., Kroeger, T., Casey, F., 2015. The role of benefit transfer in ecosystem service valuation. Ecol. Econ. 115, 51–58.
Schulp, C.J.E., Alkemade, R., 2011. Consequences of uncertainty in global-scale land cover maps for mapping ecosystem functions: an analysis of pollination efficiency. Remote Sens. 2011 (3), 2057–2075.
Schulp, C.J.E., Burkhard, B., Maes, J., Van Vliet, J., Verburg, P.H., 2014. Uncertainties in ecosystem service maps: a comparison on the European scale. PLoS One 9 (10), e109643. http://dx.doi.org/10.1371/journal.pone.0109643.
Schägner, J.P., Brander, L., Maes, J., Hartje, V., 2013. Mapping ecosystem services' values: current practice and future prospects. Ecosyst. Serv. 4, 33–46.
Scolozzi, R., Morri, E., Santolini, R., 2012. Delphi-based change assessment in ecosystem service values to support strategic spatial planning in Italian landscapes. Ecol. Indic. 21, 134–144.
Smith, J.Q., 2012. Bayesian Decision Analysis: Principles and Practice. Cambridge University Press.
Ståhl, G., Allard, A., Esseen, P.-A., Glimskär, A., Ringvall, A., Svensson, J., Sundquist, S., Christensen, P., Torell, A.G., Högström, M., Lagerqvist, K., Marklund, L., Nilsson, B., Inghe, O., 2011. National inventory of landscapes in Sweden (NILS) – scope, design and experiences from establishing a multiscale biodiversity monitoring system. Environ. Monit. Assess. 173, 579–595.
Syrbe, R.-U., Walz, U., 2012. Spatial indicators for the assessment of ecosystem services: providing, benefiting and connecting areas and landscape metrics. Ecol. Ind. 21, 80–88.
Tammi, I., Mustajärvi, K., Rasinmäki, J., 2017. Integrating spatial valuation of ecosystem services into regional planning and development. Ecosyst. Serv.
TEEB, 2010. A Quick Guide to The Economics of Ecosystems and Biodiversity for Local and Regional Policy Makers, 2010 (http://www.teebweb.org, accessed at 28.11. 2010).
Tomppo, E., Haakana, M., Katila, M., Peräsaari, J., 2008. Multi-source national forest inventory – Methods and applications. Managing Forest Ecosyst. 18, 374.
Turtiainen, M., Salo, K., Saastamoinen, O., 2011. Variations of yield and utilisation of bilberries (Vaccinium myrtillus L.) and cowberries (V. vitis-idaea L.) in Finland. Silva Fennica 45 (2), 237–251.
Van der Biest, K., Vrebos, D., Staes, J., Boerema, A., Bodí, M.B., Fransen, E., Meire, P., 2015. Evaluation of the accuracy of land-use based ecosystem service assessments for different thematic resolutions. J. Env. Manage. 156, 41–51.
Vauhkonen, J., Ruotsalainen, R., 2017. Assessing the provisioning potential of ecosystem services in a Scandinavian boreal forest: Suitability and tradeoff analyses on grid-based wall-to-wall forest inventory data. For. Ecol. Manage. 389, 272–284.
Vauhkonen, J., Packalen, T., 2017. A markov chain model for simulating wood supply from any-aged forest management based on National Forest Inventory (NFI) Data. Forests 8 (9), 307.
Vihervaara, P., Mononen, L., Auvinen, A.-P., Virkkala, R., Lu, Y., Pippuri, I., Packalen, P., Valbuena, R., Valkama, J., 2015. How to integrate remotely sensed data and biodiversity for ecosystem assessments at landscape scale. Landscape Ecol. 30, 501–516.
Vorstius, A.C., Spray, C.J., 2015. A comparison of ecosystem services mapping tools for their potential to support planning and decision-making on a local scale. Ecosyst. Serv. 15, 75–83.