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Evaluating the contribution of forest ecosystem services to societal welfare through linking dynamic ecosystem modelling with economic valuation

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ABSTRACT

Trade-offs exist among the multiple ecosystem services (ES) generated by forests. Generally, wood production conflicts with the provisioning of public-good ES such as the storage of carbon, nutrient retention and conservation of biodiversity. Recognizing that forests generate both private- and public-good ES implies that forestry should be optimized to maximize the contribution of forests to societal *welfare*. Here we develop an integrated approach for evaluating the contribution of forest ES to welfare. Our approach links the results from dynamic ecosystem modelling to economic valuation and benefit-cost analysis to evaluate the impacts of alternative forestry practices on welfare. We apply the approach to a Norway spruce forest in southern Sweden. We show that current practices are not maximizing societal welfare, because of conflicts in the optimal choice of practices from society's and forest owners' perspectives, and the distribution of welfare between generations. In particular, intensifying biomass production is shown to reduce welfare due to the concomitant degradation of public-good ES, while welfare would improve through expansion of continuous cover forestry. We anticipate that this type of approach will aid the sustainable development of forestry, by informing decision makers of the impacts of alternative forestry practices on societal welfare.

1. Introduction

Trade-offs exist among the diverse ecosystem services (ES) generated by forests (Pohjanmies et al., 2017; Eyvindson et al., 2018). Changes in forestry practices aiming to increase wood and biomass production can therefore conflict with the provisioning of public-good services that are not usually valued on markets, such as the storage and sequestration of carbon, biodiversity conservation and the quality of water originating from forest catchments (Zanchi et al., 2014; Gutsch et al., 2018). Recognizing that forests provide society with a multiplicity of values in the form of both private- and public-good ES, implies that management should be optimized to maximize the overall contribution of forests to societal welfare (Schroder et al., 2016). This necessitates considering the impacts of changes in management on all forest ES, and not just commercial products. However, since forest managers are generally individuals or companies, management is primarily driven by income and profit motives, implying that services that are not traded on markets will be largely ignored in management decisions (Lindahl et al., 2017; Bösch et al., 2018). Such circumstances create conditions for a market failure, i.e., the failure of laissez-faire to achieve the efficient or optimal management of forests from society's perspective (Bateman et al., 2013).

Information on the impacts of different forestry practices on multiple ES is urgently needed by policymakers to support decision-making to achieve environmental and sustainability goals (Kangas et al., 2018). However, as pointed out by Ninan and Inoue (2013) there is hardly any information regarding how changes in management impact the provisioning of forest ecosystem services. This is because a large number of research studies are confined to quantifying the *status* of services through the mapping of indicators (García-Nieto et al., 2013; Hansen and Malmaeus, 2016; Plas et al., 2018; Shiwei et al., 2018).

To fill this gap, emerging studies evaluate the effects of alternative forestry practices on multiple forest ES. However, these usually compare impacts based on biophysical indicators that are measured in different units and normalized to a common, but incongruent scale (Duncker et al., 2012; Gamfeldt et al., 2013; Zanchi et al., 2014). Quantifying the impacts of alternative practices on ES in terms of biophysical indicators is necessary, but not sufficient for informing societal decision-making. This is because comparing impacts measured in different units implies meeting arbitrary targets or vague goals, rather

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than achieving optimal trade-offs for the welfare of different groups and generations within society (Mendelsohn, 2019).

One approach that is suggested for systematically evaluating trade-offs is Multi-Criteria Decision Analysis (Schwenk et al., 2012). In this approach, weights are attached to different forest ES based on the relative importance of the service to the decision maker. A limitation of this approach is that the assignment of weights is highly subjective, for example based on the opinions of experts or a single decision-maker (Eggers et al., 2019). This subjectivity has serious limitations for informing societal decision making such as forest policy, because it is unclear to what extent assigned weights reflect the preferences of a particular individual, rather than citizens' preferences generally. Further, this approach ignores time preferences, which is a fundamental consideration when evaluating impacts on societal welfare, because society includes even future generations (Arrow et al., 2013).

Ideally, policymakers need to know how and when potential changes in the flows of different services impact the welfare of different groups within society when trade-offs are unavoidable. A powerful form of decision support that takes a societal perspective and explicitly considers time preferences is Benefit-Cost Analysis (BCA), which is grounded in Welfare Economic Theory, the purpose of which is determining the desirability of policy proposals (Johansson, 1991). That is, BCA of a policy intervention forces the decision-maker to systematically consider who the beneficiaries and losers are among different groups in society, and not just forest owners, and between current and future generations (Hanley and Spash, 1993).

In short, BCA of changes in flows of forest ES involves translating changes in biophysical indicators of different ES (i.e., the actual service) to a common denominator of value, which is usually expressed in monetary terms (OECD, 2018). Monetary valuation is however more than money per se, but is an indicator of the amount of income that citizens forming a society would be willing to give up in order to obtain an additional unit of a particular forest ES, without becoming worse off. This framing recognises that society has limited resources (incomes) and that we are obligated to choose among competing alternatives for these resources (e.g., education, healthcare, etc.). Consequently, the marginal value of a public-good ES to society can be understood as being equivalent to a variation in individual incomes. Market prices of forest products have this characteristic, but as we shall show, approximations of value can also be found for public-good ES that do not have market prices (Atkinson et al., 2012).

We do not suppose that such a translation to a monetary measure is the same as the welfare generated by a particular service, but monetary valuation on the margin (i.e., for non-extreme changes in ES) can provide a good approximation of the potential change in societal welfare associated with different forestry practices (Just et al., 2004). Thus, we caution that our approach is intended for evaluating noncritical or marginal changes in natural capital and associated ecosystem services, such as brought about by incremental changes in the management of large areas of homogenous production forests as studied here, rather than, e.g., the preservation or not of an old-growth forest that has priceless intrinsic value.

As far as we know, no other studies combine dynamic simulation of the effects of alternative forestry practices on forest ES with economic valuation and BCA, but other forms of economic valuation exist. For instance Bösch et al. (2017) adopt an economic approach based on input-output modelling, and valuation based on changes in GDP rather than welfare economic theory, while Häyhä et al. (2015) combine valuation with mapping to determine the total economic value of Alpine forests, and Ninan and Kontoleon (2016) estimate the total economic value of a forest in Japan and India respectively. The approach we advocate is to focus on valuing differences or value-added in the supply of ES generated by different management practices. This is sufficient, given regard to the precaution called for above, to identify optimal strategies to maximize the total contribution of a forest to societal welfare (by the principles of mathematical optimization), and to inform the design of policy instruments that increase the societal value of forests. Finally, rewarding forest owners for value added rather than total value is also important for designing efficient payments for ecosystem services, since conservation budgets are limited (Fortmann et al., 2016).

Models are commonly used to assess the impacts of alternative management practices on forest ES, because experiments that compare management alternatives are limited and difficult to upscale, due to the heterogeneity and complexity of forest ecosystems (Kangas et al., 2018). A further advantage is that models can project effects into the future while considering environmental changes such as climate change (Lagergren and Jönsson, 2017).

This explorative study aims to evaluate the contribution of alternative forestry practices and associated flows of ES to societal welfare, through linking dynamic ecosystem modelling with economic valuation of changes in flows of forest ES and BCA under alternative climate scenarios. We apply a specific dynamic ecosystem model to illustrate a general approach for utilizing output from ecosystem models as input to welfare evaluation, since model output indicators are generally similar. To exemplify the approach, we applied it to a virtual forest area in southern Sweden. Thereafter we present the results of the evaluation in conjunction with discussion to aid interpretation of the results, and finally our conclusions.

2. Materials and methods

In this section we begin by describing the study area and motivating the selection of forest ES evaluated in this study, and the choices of indicators used to measure simulated changes in flows of the selected services. We then introduce the ecosystem model, the different management practices evaluated and an alternative climate scenario that is used to test the sensitivity of the results to predicted climate change. Thereafter we describe the welfare economic principles and data used to derive marginal values of the different forest ES, and finally the benefit-cost analysis used to aggregate changes in the values of the flows of the different ES over time.

2.1. Study area

The study area is a virtual managed forest dominated by Norway spruce in southern Sweden that has a rotation period of 70 years under the current management system. The forest area is divided into a chronosequence of 70 patches of one hectare each with identical properties, but with different stand ages, ranging from 1 to 70 years to mimic a complete rotation. It is assumed that each simulated patch has the properties of a real spruce forest site in southern Sweden, Västra Torup (13.51E, 56.14N). The site is relatively flat on a brown podzolic soil and has an average annual temperature of ~8 °C and annual precipitation of 900 mm. Such spruce forest accounts for 2.0 million ha or ~40% of the productive forest area in southern Sweden, i.e., Götaland (SLU, 2019). This site has previously been described and used to assess the effects of forest management intensification on multiple ecosystem indicators in Zanchi et al. (2014). Details of the representative forest patch forming the basis of the chronosequence in terms of input data and model validation are reported in the Supplementary Material.

The ensuing results are presented as average values per unit of area (ha) for the entire virtual study area (70 ha) at the end of the simulation period (2100). By considering the average value in the rotation system per unit of area, the results are independent of the extent of the study area and can be considered representative for spruce forests with similar landscape properties and rotation systems.

2.2. Identification of indicators

We identified a set of five indicators representative of key forest ES in the study area (Table 1) that can be simulated by the dynamic

Table 1
Ecosystem indicators and corresponding ecosystem services investigated in this study.

Indicator	Acronym	Unit of measure	Ecosystem service
Harvested biomass	Harv	g d.m. m^{-2} y^{-1}	Production of raw materials
Tree carbon stock	TreeC	g C m^{-2}	Climate regulation
Soil carbon stock (topsoil)	SOC	g C m^{-2}	Climate regulation
Nitrogen leaching	Nleach	g N m^{-2} y^{-1}	N retention (Water quality)
Deadwood biomass	DeadW	g d.m. m^{-2}	Biodiversity

Notes: d.m. is dry matter, C is carbon and N is nitrogen.

ecosystem model ForSAFE (Section 2.3.1) and can be used to value changes in forest ES (Section 2.4). An additional and potentially key indicator related to water quality simulated by the model is pH; however, since this indicator was not significantly affected by any of the management or climate scenarios we omitted it from the results. Other potentially relevant ES for the site, such as non-timber forest products (e.g., game and mushrooms), and recreational services could not be included in this study because currently they are not represented by an indicator simulated by ForSAFE. From an economics perspective, production of raw materials is a private good whereas the other four selected services are public goods, the implications of which are made clear in Section 2.4.

2.3. Dynamic ecosystem modelling and analysis

We applied the dynamic ecosystem model ForSAFE to assess the effects of alternative forest management practices on the five ecosystem-service indicators in the virtual forest area in southern Sweden.

2.3.1. ForSAFE model

The ForSAFE model simulates the dynamic responses of forest ecosystems to environmental change (Fig. 1). It is a mechanistic model based on the aggregation of interacting biogeochemical processes (Wallman et al., 2005; Belyazid et al., 2006) and combines the engines of four established models: the tree growth model PnET (Aber and Federer, 1992), the soil chemistry model SAFE (Alveteg, 1998), the decomposition model Decomp (Walse et al., 1998; Wallman et al., 2006) and the hydrology model PULSE (Lindström and Gardelin, 1992). Simulated tree growth is driven by temperature and solar radiation. This potential growth is constrained by water and nutrient availability, while nutrient availability is determined by decomposition, deposition and mineral weathering. Tree growth is further affected by disturbances, such as forest management and storms. The mechanisms employed in the ForSAFE model are generic and therefore the model is suitable for application to different forestry conditions and climatic regions (Sverdrup and Belyazid, 2015).

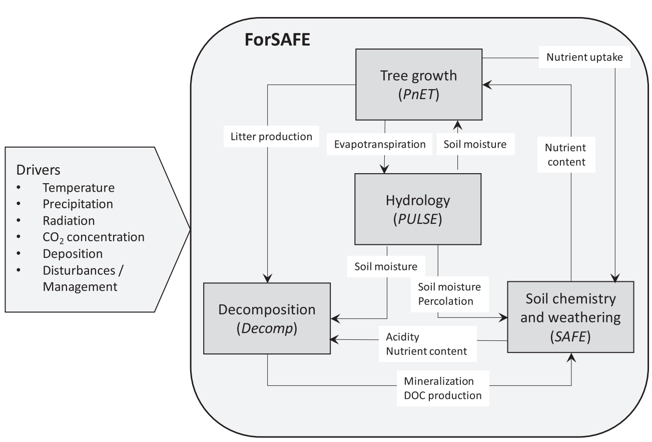

Fig. 1. Illustration of the different components of the ForSAFE model and their interactions. The names of the original models combined in ForSAFE are reported in brackets. The factors driving model processes are listed to the left of the figure.

2.3.2. Management scenarios

The ForSAFE model was applied to simulate the effects of different management practices on the selected ecosystem-service indicators (Table 1). We considered a benchmark and three alternative management scenarios that are relevant for the study area:

i. Business-as-usual (**BAU**): the current forest management system that consists of clear-felling after a rotation period of 70 years is the benchmark scenario. During the rotation the forest is thinned twice (at 25 and 45 years). Only tree stems are removed from the site after the clear-felling and thinnings, while harvest residues are left on site.
ii. Whole tree harvesting (**WTH**): an identical harvesting regime to the BAU scenario, but stems and residues (tops, branches and 60% of the foliage) are also removed from the forest after the final clear-felling and intermittent thinnings.
iii. Whole tree harvesting with shorter rotation (**WTH + SR**): the rotation period is shortened to 60 years, and stems and residues are removed from the forest after the clear-felling and two thinnings (at 20 and 35 years).
iv. Continuous cover forestry (**CCF**): the canopy cover is continuously maintained by removing only a fraction of the tree biomass (15%) through frequent harvesting (every 10 years), thereby avoiding clear-felling; it is also assumed that only stems are removed from the site. By assuming this harvesting system, the mean annual biomass increment is ~83% of that in the BAU scenario (i.e., 3.96 t ha^{-1} yr^{-1} d.m. compared to 4.75 t ha^{-1} yr^{-1} of d.m. over the 70 year BAU rotation period), which is comparable to data previously reported (Lundmark et al., 2016).

The model was applied to simulate the four management scenarios in the virtual study area. The real forest site in Västra Torup was clear-felled in 2010. Therefore, it is assumed that alternative management scenarios are applied after this event and diverge from the BAU scenario at the first thinning (2040 in WTH, 2030 in WTH + SR and 2020 in CCF). Model results were replicated for each forest patch included in the study area, according to the age of the forest patch in the chronosequence . The average indicator values for the entire study area were used to assess the effects of the alternative management scenarios on the ecosystem-service indicators by contrasting them in spider diagrams. Since the scales of the indicators differ by large orders of magnitude, we present the relative changes in indicator values compared to the business-as-usual scenario (Zanchi et al., 2014), i.e.:

$$\Delta I(t) = \frac{I_{ALT}(t) - I_{BAU}(t)}{|I_{BAU}(t)|} \times 100 \quad (1)$$

where $\Delta I(t)$ is the percentage change in a specific indicator in year t (e.g., TreeC) and I_{ALT} and I_{BAU} are the values of the indicator under the alternative and business-as-usual scenarios in year t. We present the indicator changes at the end of the simulation period (t = 2100).

2.3.3. Climate scenarios

Climate data are based on simulations of the Global Climate Model ECHAM (Roeckner et al., 2003) that has been calibrated with historical climate data. These data have been derived from the nearest Swedish Bureau of Meteorology (SMHI) weather station using spatial interpolation and from the NCEP/NCAR Reanalysis project for solar radiation (1961–2008) (David Rayner, 2010, personal communication).

The effects of the different management scenarios on the service indicators were simulated under two future climate scenarios:

- Reference climate scenario (**ClimRef**): is based on historical weather by repeating observed temperatures and precipitation from 1981–2010 to 2100.
- Climate change scenario (**ClimCh**): follows the SRESA2 emissions story line of the IPCC for the period 2010–2100.

2.4. Economic valuation of changes in forest ecosystem services

In this section, we describe the methods used for valuing the simulated changes in forest ES. Since market prices do not exist for the public-good services, their assumed marginal values are based on various nonmarket valuation techniques (Atkinson et al., 2012) used to value these particular services in Sweden. As such our assumed values are based on Swedish citizens' preferences to be consistent with principles of welfare economic analysis at the national level.

It is important to realise that the purpose of economic valuation is to inform decision-making, thus an assumption about the decision-maker's objective must be made. The decision problem we investigate is whether it would increase societal welfare if one hectare of the studied forest area, all other things being equal, was managed according to one of the three alternative management scenarios. Such a small change would not be sufficient to influence the derived marginal valuations of forest ES. In this respect, we illustrate the principles of economic valuation for informing decision-making, but do not answer the question as to what the optimal area of each particular management scenario would be. Such an exercise would require complementary modelling that considers market effects on the marginal values of the different services (e.g., a large change in the timber harvest might cause a change in the price of timber). In this evaluation, we assume therefore that the quantitative changes in the levels of forest ES are small relative to the total levels generated by Swedish forests, and hence marginal values are held constant in all simulations.

Finally, we calculated the Net Present Value of future changes in services to take into account peoples' time preferences, i.e., bearing in mind that benefits occurring in the future are less valuable than a benefit occurring today (Arrow et al., 2013). We apply a standard approach and social rate of discount, but because this choice is not fully objective, we also subject it to sensitivity analysis and discuss the implications of alternative forms of discounting.

2.4.1. Forest stem-wood products and residues

Harvested wood and residues are private goods and therefore conducive to valuation on markets. The Swedish forestry sector is largely free from government subsidies and boarder protection; hence prices faced by Swedish foresters are generally world market prices. We therefore use the weighted-average annual price of Swedish timber production over the period 2010–16 to approximate the marginal societal value of Swedish forestry products (Table 2).

The valuation of forest products must also consider differences in harvesting costs in the alternative scenarios. In particular, there could be a substantial difference in the unit costs of harvesting timber using clear-felling compared to continuous cover forestry (CCF). CCF is rare and hence not represented in Swedish forestry statistics. The unit costs of CCF have though been found in forestry experiments to be similar to those of thinning and around 30% higher than clear-felling (Hannerz et al., 2017, p. 38), because both require selective harvesting of individual stems. Consequently, we assume that thinning and CCF have a unit harvest cost of 140 SEK per m^3 wood or 0.35 SEK per kg d.m. (i.e., 0.27 × 1.3 = 0.35, as per Table 2).

Table 2
Prices and harvesting costs of forest products, weighted-averages 2012–16.

	Quantity ($M\ m^3$)	Revenues (Costs) (M SEK)	Price (Cost) (SEK/m^3 or kg)
Revenues			
Stem-wood products			
- Timber	39.24	16 255	414.23
- Pulp for paper	35.10	8 733	248.80
- Other timber	0.60	307	511.60
- Firewood	7.22	1 117	154.69
*Weighted average stem-wood price (SEK/kg)**			**0.80**
Residues			
- Branches and tops	3.56	666	187.01
Weighted average residue price (SEK/m^3)			187.01
*Weighted average residue price (SEK/kg)**			**0.47**
Harvesting costs			
Clear-cutting stemwood		(9 005)	(107.68)
*Weighted average cost Stem-wood (SEK/kg)**			**(0.27)**
Residues		(425)	(119.29)
*Weighted average cost Residues (SEK/kg)**			**(0.30)**

* To convert harvested quantities from m^3 to kg of wood we used a wood density of 400 kg per m^3 (IPCC, 2006). *Source*: The Swedish Forest Agency's Statistical Database (SFA, 2017) table "Calculated volume and monetary value of annual fellings in Sweden (in current prices) by Items, Assortment and Year.").

2.4.2. Climate regulation: Carbon stocks in trees and the topsoil

Estimating the marginal benefits to Swedish citizens of reducing greenhouse gas emissions is challenging. First, climate change is a global problem and second, the future costs of climate change and hence the potential benefits of reducing emissions today are highly uncertain. The Swedish parliament has responded by legislating a carbon tax, starting in 1991 at a rate of 250 SEK/t CO_2e, and successively increasing the tax over time to its current rate of 1180 SEK/t CO_2e or 136 USD/t CO_2e (Government Offices of Sweden, 2019).[1] Since this tax rate has emerged from the Swedish parliament as a product of political deliberation and has been in place for almost three decades it can be argued that it is a stable reflection of Swedish citizens' preferences (De Nocker et al., 2004). Although relatively high on world standards, it is in the vicinity of the range of rates that have been previously estimated for a globally optimal carbon tax of 40–100 USD/t CO_2e (Warren, 2014). More recently, Crost and Traeger (2014) find that the optimal tax rate doubles when considering the uncertainty of future damages. Finally, the UN Global Compact argues that 100 USD/t CO_2e is the minimum tax needed to achieve the 1.5–2-degree Celsius pathway (UN, 2016).

Given the consensus that a global tax rate in the range of 40–200 USD/t CO_2e is needed to avoid climate change, and that the rate set by the Swedish parliament falls within this range, we adopt the Swedish tax as the marginal value to Swedish citizens of the forest ecosystem service climate regulation.

2.4.3. Nutrient retention: avoided nitrogen pollution of Baltic Sea

A major environmental problem related to nutrient emissions from Swedish lands to water is eutrophication of the Baltic Sea (HELCOM, 2018). A goal of the HELCOM Baltic Sea Action Plan is a sea unaffected by eutrophication, which is currently so severe that it is causing the wide-scale incidence of dead-sea bottoms (*ibid.*). Increasing retention of

[1] Converted to USD using the average exchange rate of 8.70 SEK/USD in 2018. Source: Sveriges riksbank. Search interest & exchange rates from https://www.riksbank.se/en-gb/statistics/search-interest–exchange-rates/.

nutrients by Swedish forests provides therefore an ecosystem service in the form of better water quality. Ahlvik and Ahtiainen (2014, p. 13) have derived marginal values of improving Baltic Sea water quality. They estimate the benefits to each of the nine coastal states of the Baltic Sea of reducing one kilogram of nitrogen (and phosphorous) to the seven different basins of the sea from their current, poor quality. The impacts on water quality are determined by a basin level marine model (Ahlvik et al., 2014) and the marginal benefit estimates derived from 10,500 responses to a Contingent Valuation survey conducted across the nine states, where incidentally Swedes were found to have the highest willingness-to-pay to improve water quality (Ahtiainen et al., 2014). Since our study area is in southern Sweden we adopt their estimate of 11.46 EURO/kg N or 124.19 SEK/kg N of the marginal benefits to Swedish citizens of improving water quality in the Baltic Sea Proper Basin to which runoff from our study area flows.[2]

Nutrient leaching also impacts the quality of inland waters, however, due to lack of relevant valuation studies we ignore this potential impact, which implies our evaluation risks undervaluing the contribution of nutrient retention services to societal welfare.

2.4.4. Biodiversity: deadwood

The volume of deadwood is a key indicator of biodiversity in forests (Rondeux and Sanchez, 2010; Gao et al., 2015) and is listed by Forest Europe among the pan-European indicators for sustainable forest management under the criterion "Maintenance, Conservation and Appropriate Enhancement of Biological Diversity in Forest Ecosystems" (Forest Europe, 2015). It is also recognised that the volume of deadwood must increase by 40% to maintain biodiversity in Swedish production forests (Jong and Almstedt, 2005), a level which has also been formalised by the Swedish Parliament (2001) within the National Environmental Objective, Healthy Forests (Swedish EPA, 2008a). The volume of deadwood can also be dynamically simulated based on tree mortality rates, and therefore it was chosen as a relevant indicator to assess biodiversity values with ForSAFE.

It is naturally very challenging to put a monetary value on conserving biodiversity, because available techniques and studies provide an incomplete perspective on it (Nunes and van den Bergh, 2001). Still the impacts of forest management on biodiversity are crucial to achieving sustainability goals. We therefore make an assumption about the value of leaving deadwood in Swedish production forests based on a Choice Experiment commissioned by the Swedish EPA (2008b) that was sent to 1000 randomly selected Swedes between the ages of 18–75, to determine the average Swedish households' willingness-to-pay (WTP) "to take actions to protect all threatened forest plant and animal species" compared to the current situation. We therefore interpret this valuation as Swedish citizens' WTP to protect biodiversity per se (a cultural ES), because the choice experiment is confined to threatened species. Using the approach detailed in the Supplementary Material, we derived an equivalent, average annual WTP by Swedish households to increase the stock of deadwood in production forests, which is 1.11 SEK/kg change in the stock of dead wood, assuming that dead wood has a lifespan of 67 years as habitat (Ranius and Kindvall, 2004).

To test the sensitivity of the results to changes in the uncertain valuations of the public-good ES, we vary the assumed valuations by ± 20% in a sensitivity analysis.

2.5. Welfare economic analysis of changes in forest ecosystem services

For each annual time step t in the simulation period T, we value the change in the indicator for a particular forest ecosystem service i as

$$\Delta V_{i,t} = P_i(Q_{i,t} - Q_{i,t-1}) \tag{2}$$

where P_i is the marginal value of ecosystem service $i \in$ (Harv, TreeC, SOC, Nleach, DeadW) and $Q_{i,t}$ is the quantity or level of a forest ecosystem service i as measured by its indicator at time $t=(0,T)$. Accordingly, the term in brackets is the change in the level of the service compared to the previous year.

Aggregating changes in value over time presents a similar dilemma to comparing changes in services with different indicators (units), because benefits occurring in the future are less valuable to people than a benefit occurring today due to discounting (Gollier, 2011). To make values comparable over time it is necessary to calculate the present value (PV) of each change in value before aggregating across years. The standard approach to calculating the PV of a future value, in this case the change in value of a particular ecosystem service $\Delta V_{i,t}$, is:

$$PV_{i,t} = \frac{\Delta V_{i,t}}{(1+r)^t} \tag{3}$$

where r is the discount rate.

It follows that the Net Present Value of changes in all forest ES is a sum over all services and all time periods discounted to their present values:

$$NPV = \sum_{i=1}^{I}\sum_{t=1}^{T}\frac{\Delta V_{i,t}}{(1+r)^t}. \tag{5}$$

To illustrate the implications of discounting, the present value of 1 SEK for t = 50 and r = 3% is 0.23 SEK, which is clearly a substantially lower value than if the benefit was realized today. Since the choice of discount rate is to some extent subjective, we present the results for a range of discount rates.

When evaluating forestry from society's perspective it is necessary to use the social rate of discount, which will usually be lower than private rates of discount. A standard choice for the social rate of discount in Sweden is 3–3.5% which is motivated for standard investments such as roads (Svensson and Hultkrantz, 2014). However, it has been argued that the long periods associated with climate change and its risks for future generations, imply that standard forms of discounting are unfair. Instead it is argued that projects longer than 30 years should be evaluated using lower rates of discount (Sterner and Persson, 2008) and that projects in the far-distant future (centuries) should be discounted at the lowest possible rate using declining or hyperbolic discounting (Weitzman, 1998). Given the moderate time-frame evaluated here, we adopt a standard exponential approach to discounting and a social discount rate of 2% as our Baseline rate, i.e., a rate lower than the standard rate. To test the sensitivity of the results to this choice we also test an extremely low rate of 1%, and a higher or standard rate of 3%. In principal, the lower rate accords greater weight to the welfare of future generations while the higher rate prioritizes current generations' welfare, because discounting downgrades costs and benefits occurring in the future. This range of choices should be sufficient to gauge the implications of the choice of discount rate for the results.

3. Results and discussion

3.1. Dynamic ecosystem modelling

To begin with, we simulated the effects of all four management scenarios on the ecosystem indicators for the virtual study area (i.e., average levels for the chronosequence) under the reference climate scenario (Fig. 2), after which we tested the sensitivity of the results to the climate change scenario (Table 3).

The purpose of more intensive forest management practices is to increase biomass production (Verkerk et al., 2011a). The model results confirmed that the more intensive WTH and WTH + SR scenarios progressively increased the total amount of wood extracted from the simulated forest (Lundmark et al., 2014). Conversely, cumulative wood

[2] Converted to SEK using the average exchange rate of 10.26 SEK/EURO in 2018 and 2018 value after adjusting for an average chore inflation rate (CPIF excl. energy and unprocessed food) of 1.38 % p.a. over the four-year period 2015–18. Source: *ibid.*

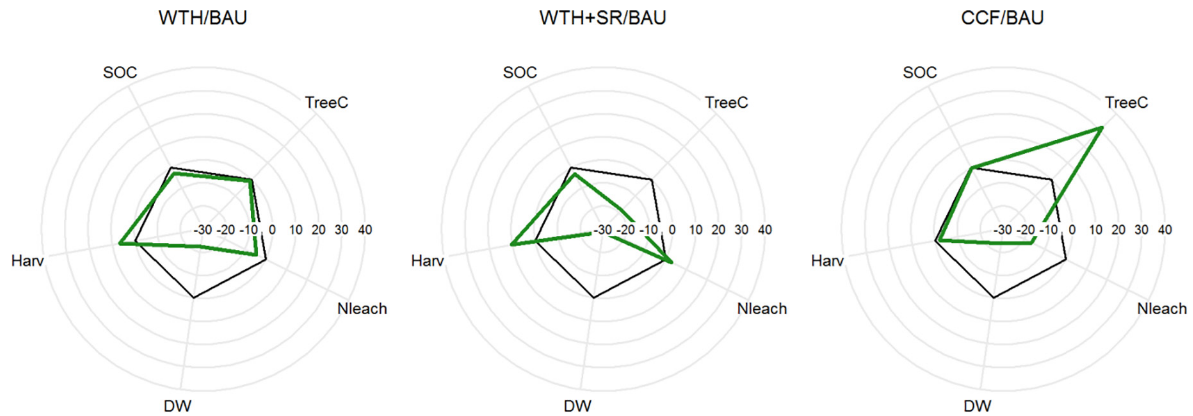

Fig. 2. Integrated assessment of changes in the ecosystem-service indicators under the three alternative management practices as percentage changes compared to the business-as-usual scenario (BAU) in 2100 (WTH: whole tree harvesting; WTH + SR: whole tree harvesting and shorter rotation period; CCF: continuous cover forestry; SOC: soil organic matter; TreeC: tree carbon; Nleach: nitrogen leaching; DW: deadwood; Harv: harvested wood).

Table 3
Simulated changes in forest ES indicators under the climate change scenario (ClimCh) for the different scenarios compared to the benchmark scenario in 2100 (i.e., BAU under the reference climate scenario (ClimRef), e.g., $(WTH^{ClimCh} - BAU^{ClimCh})/BAU^{ClimRef}$.

Indicator	Absolute Changes				Change (%)	Additive change compared to BAU (%)		
	BAU	WTH	WTH + SR	CCF	BAU	WTH	WTH + SR	CCF
TreeC (t ha^{-1})	−13.3	−12.8	−10.4	−19.7	−17.2%	0.4%	0.7%	−2.2%
SOC (t ha^{-1})	−1.3	−1.1	−1.1	−1.2	−2.4%	0.4%	0.4%	0.1%
Harv (t ha^{-1})	−16.5	−21.4	−22.4	−10.8	−2.8%	−0.6%	−0.7%	0.9%
DW (t ha^{-1})	−8.6	−6.3	−5.4	−6.3	−15.8%	0.8%	1.8%	0.7%
Nleach (kg ha^{-1})	20.9	10.8	2.9	29.9	6.4%	−2.9%	−5.5%	4.5%

production in CCF was slightly lower but comparable to the production in the BAU scenario; however Peura et al. (2018) have previously reported a greater harvest reduction in CCF compared to clear-felling. The difference in results is most likely due to different assumptions about the type and intensity of management practices in the two scenarios and the higher productivity of forests in southern Sweden.

In agreement with previous studies, trade-offs occurred between wood production and public-good services (Pohjanmies et al., 2017). The tree-carbon stock declined substantially due to the introduction of a shorter rotation with clear-felling (WTH + SR); whereas the elimination of clear-felling with CCF substantially increased tree-carbon stocks. In both cases, the change in carbon stocks was related to the progressive reduction or accumulation of tree biomass in the respective scenario (Schwenk et al., 2012). The higher tree C stock in CCF is in line with results in Peura et al. (2018) who, however, reported a smaller difference between tree carbon stock in CCF and BAU scenarios (about 8%). Residue extraction did not affect tree biomass, and thereby tree carbon, within the considered period as reported by previous research (Repo et al., 2015). However, over longer periods, residue extraction could negatively affect nutrient availability in nutrient limited forest soils and, ultimately, reduce tree growth and biomass (Ranius et al., 2018).

Soil carbon stock in CCF was comparable to the BAU scenario, while it declined slightly in WTH and WTH + SR. The most substantial reductions being generated by the most intensive management scenario (WTH + SR) through reduced litter inputs due to the removal of residues and lower tree biomass. Nevertheless, soil C stock changes were small, confirming that the effects on soil carbon are not clear and might be marginal or absent in the alternative scenarios (Clarke et al., 2015; Ranius et al., 2018).

Moreover, all alternative management scenarios generated long-term reductions in deadwood stocks (Verkerk et al., 2011b; Peura et al., 2018). The reduction in deadwood stock was caused by less biomass being left at the site after clear-felling under WTH and WTH + SR and the absence of clear-felling and thereby of large deadwood inputs under CCF.

Finally, intensification of forestry had a limited effect on nitrogen leaching compared to the BAU scenario, with slightly higher nitrogen leaching in WTH + SR and slightly lower in WTH, which confirms that the effects on nitrogen export can be variable (Ranius et al., 2018). In contrast, CCF resulted in substantial long-term reductions in nitrogen leaching by avoiding the peaks of nitrogen leaching caused by clear-felling (Akselsson et al., 2004).

The integrated assessment of indicator changes using spider diagrams (Fig. 2) shows substantial effects of the alternative management practices on tree carbon and deadwood stocks. It also shows a significant reduction in nitrogen leaching under CCF. However, the comparison fails to highlight the changes in terms of wood production, which are very relevant in absolute terms because it affects foresters' incomes, but less evident in terms of percentage change. This exemplifies the limitations of basing decisions on changes in the indicators alone when there are trade-offs and scale differences, and hence the need for valuation. That is, as each of the alternative scenarios can be seen to have advantages and disadvantages compared to BAU on the generation of different forest ES, valuation is necessary to determine an optimal management.

As a sensitivity test, we included climate change in the analysis (Table 3) and the model projected systematic negative effects on all forest ES. A further decline of tree carbon stock was simulated in the future under all scenarios due to reduced water availability during the growing season. Further, the soil carbon stock increased less than under the reference climate scenario, while wood production and deadwood stock were lower, and more nitrogen leached from the soil.

A current limitation of using dynamic models such as ForSAFE to assess forest ES under alternative scenarios is the lack of indicators representing recreational and cultural services and the provision of non-timber products such as game and mushrooms. These are services that are usually assessed through the analysis of qualitative or statistical information (Gamfeldt et al., 2013; Hermes et al., 2018) and are not

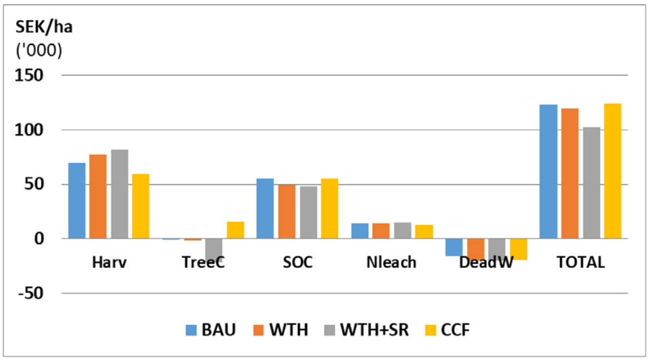

Fig. 3. Welfare economic impacts of the different forestry management scenarios shown as present values of the simulated changes in forest ecosystem services and in total for the period 2010–2100, given the assumed social discount rate of 2% and marginal valuations of forest services (Section 2.4).

linked to physical properties of the forests that can be simulated dynamically by ecosystem models. However, innovative techniques, such as linking recreational values to landscape characteristics (Hermes et al., 2018; Van Berkel et al., 2018), are emerging and could be used in the future to identify indicators that can be dynamically modelled.

3.2. Benefit-cost analysis

We begin by presenting the BCA results given the assumed marginal valuations of the selected forest ES (Section 2.4) and testing their sensitivity to the choice of discount rate. Thereafter we present a sensitivity analysis of the baseline values of the public-good ES using Monte Carlo Simulation (for method see Supplementary Material).

3.2.1. Valuation results for various discount rates

The valuation of changes in forest ES indicates that increasing the area of CCF would make a higher contribution to societal welfare (when applying the baseline social rate of discount) than continuing with BAU or switching to WTH or WTH + SR on all the area (Total in Fig. 3). On the one hand, if we ignore the value of public-good services and focus on the harvest value alone (Harv), then CCF would be the least desirable alternative and WTH and WTH + SR clearly the best, because the latter scenarios result in the greatest output of commercial forest products. On the other hand, WTH and WTH + SR provide the lowest total marginal contribution to societal welfare, because they generally provide the least marginal benefits in the form of the public-good ecosytem services. This is because the greater extraction of biomass in these scenarios, through the extraction of residues, as well as the shorter rotation period in WTH + SR, have a substantial societal cost in the form of lower carbon storage in the forest (TreeC plus SOC) and biodiversity value (deadwood stocks). However, in this study we do not consider the carbon stored in harvested wood products and the carbon benefits from the substitution of fossil fuels. If these were included in the carbon balance, the carbon benefits of WTH and WTH + SR would be higher (Lundmark et al., 2016) and their marginal contribution to societal welfare would increase.

Each of the alternative scenarios result in even lower values of deadwood conservation compared to BAU (i.e., its value also declines in BAU). This is not surprising for the WTH and WTH + R scenarios because these greatly increase wood extraction, however even CCF performs worse than BAU. This seems consistent with related research which implies that deadwood alone is too coarse an indicator for measuring the complexity of biodiversity, and that biodiversity conservation could be improved in any of the scenarios if coupled with targeted conservation interventions (Felton et al., 2016).

The choice of discount rate was important for the overall ranking of the different management scenarios, because it had a substantial effect

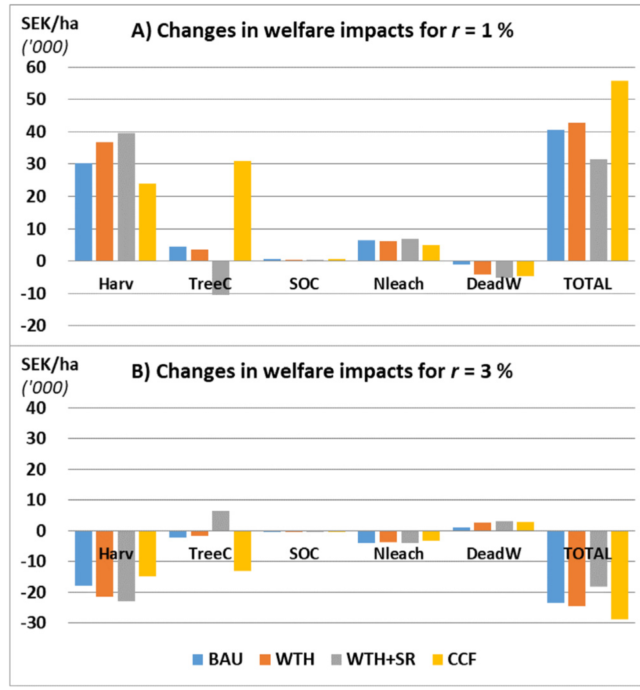

Fig. 4. Sensitivity of the present values of the simulated changes in forest ecosystem services shown in Fig. 3 to different assumptions about the social rate of discount: panel A) a lower rate of 1% and B) a higher rate of 3%.

on the relative contribution of each scenario to societal welfare (Fig. 4). The lower the discount rate (Fig. 4A), the greater the marginal benefits to society of switching to CCF on the margin. This is primarily attributable to the greater weight accorded to climate regulation through higher carbon storage in standing trees, where the main benefits occur quite far into the future. In contrast, a higher discount rate is unfavourable to CCF (Fig. 4B) because it significantly reduces the welfare value of tree carbon stocks, i.e., a higher discount rate reduces the weight of this benefit in the PV calculation. Finally, the higher rate of discount also reduces the present value of the wood harvest across all scenarios and vice versa, because forests have long rotation periods.

Overall a higher discount rate reduces the benefits of the alternative scenarios compared to BAU, and for even higher rates of discount (that might be applied by forest owners) it becomes clearly the most attractive scenario, which is consistent with current behaviour of forest managers (Eggers et al., 2019). Consequently there is likely to be a conflict in the optimal choice of management practices from society's perspective if private owners place much greater importance on current harvests than the welfare of future generations, or policymakers focus on a single issue, such as increasing the extraction of residues for biofuel feedstock in the hope of combating climate change. This illustrates the necessity to balance the provisioning of private and public-good ES as affected by forestry management between current and future generations, if the societal value of forests is to be maximized. Although the issue of discounting is contentious, it makes an otherwise implicit choice by the decision-maker explicit, and thus possible to evaluate the implied tradeoffs in welfare between generations, which is less obvious in other methodologies such as MCDA (e.g., Schwenk et al., 2012).

3.2.2. Sensitivity of results to uncertain valuations of public-good services

Overall, the rankings of the BAU, WTR and CCF scenarios are fairly sensitive to changes in the uncertain valuations of public-good services (the boxes and whiskers for Total_NPV in Fig. 5 tend to overlap for these scenarios), whereas WTH + SR is clearly the least desirable scenario since its distribution for Total NPV lies almost exclusively below the others. In particular, the valuation of TreeC is driving the ranking of the

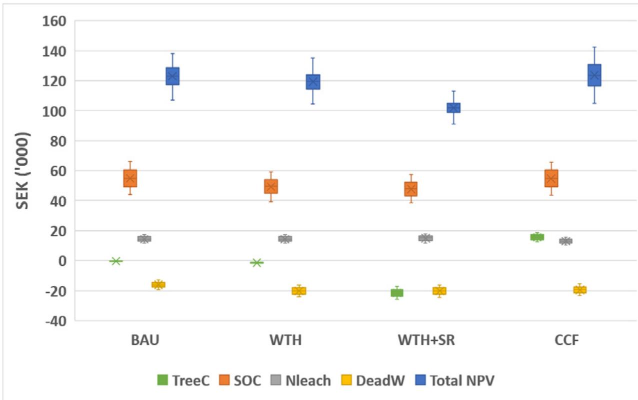

Fig. 5. Sensitivity of the present values of the simulated changes in forest ecosystem services shown in Fig. 3 to the uncertain valuations of the public-good ecosystem services. *Note:* The valuations of marginal changes in the public-good services are varied by ± 20% using Monte Carlo simulations (Supplementary Material). Sensitivity to the valuation of harvests is not tested, because forest products are valued using objective market prices, but are included in the calculation of Total NPV.

scenarios. Hence it is important for Swedish policymakers to be clear about their valuation of the climate benefits of forestry. The Swedish carbon tax on petrol, does however, provide a very clear signal that the value of the potential climate benefits of forestry is high, because GHG emissions are a global environmental problem and hence the benefits of reducing emissions are identical between sectors. That is, there is no reason to value the climate benefits of forestry differently to that of reducing the consumption of fossil fuels.

Naturally, more radical changes in any of the public-good valuations from the assumed valuations would have the potential to change the conclusions. For example, if the values of all public-good services were set to zero, then obviously only the value of the tree harvest would represent each scenarios contribution to societal welfare, in which case WTH and WTH + SR would, respectively, provide the highest marginal contributions to welfare (Fig. 3). In summary, it is crucial to understand how we have derived the assumed valuations of the public-good services (Section 2.4), because these do not have observable values such as wood products (i.e., market prices). They are, however, based on valuation methods anchored in the expression of Swedish citizens' preferences and willingness-to-pay for forest ES, and not our own preferences or those of other "experts" who might not necessarily reflect the preferences of society in general.

4. Final discussion and conclusions

This study develops an approach for evaluating the incremental contribution of both private- and public-good forest ES to societal welfare by linking the results from dynamic ecosystem modelling with nonmarket valuations of public-good ES and Benefit-Cost Analysis (BCA); thereby making it possible to evaluate the effects of implementing alternative forestry practices on the contribution of forests to societal welfare. We demonstrate the approach through an application to the management of a Norway spruce forest in southern Sweden. While the quantitative results are applicable to the conditions of the virtual study area, the mechanisms of the ecosystem model ForSAFE applied here are general and therefore applicable to a wide spectrum of forestry conditions and climates. Further, the valuation of public-good ecosystem services reflects the willingness-to-pay by Swedish citizens, as BCA is usually performed on a national basis. Nevertheless we believe our conclusions about the implications of valuing public-good forest ES and discounting have general applicability, because these values are usually ignored in commercial forestry.

Our dynamic ecosystem-modelling results corroborate previous research showing that trade-offs exist between increasing wood supply and the provisioning of public-good ES from forests (Pohjanmies et al., 2017). In particular, trade-offs arising from the intensification of forestry production through residual withdrawal and a shorter rotation period. However, our study does not consider the carbon benefits from the substitution of fossil-based materials, which could increase the benefits from more intensive management practices (Lundmark et al., 2016). Further, we use a simple indicator of biodiversity conservation value; changes in the stock of deadwood. However, forest ecosystems are complex and BCA should be complemented with additional information about potential impacts of different practices, particularly targeted measures, on biodiversity (Felton et al., 2016). Finally we only value a selection of public-good services, which likely undervalues the full contribution of public-good forest ES to societal welfare as determined here.

Visualization of the ecosystem modelling outputs illustrates the trade-offs among forest ES, but simultaneously demonstrates the limitations of basing an evaluation on indicators of ES quantified in different units and scales, because direct comparison of the total contribution to societal welfare is not possible. One way of overcoming this comparability problem is MCDA (Schwenk et al., 2012; Eggers et al., 2019), but the assignment of ES weights is subjective (e.g., based on expert opinion) and time-preferences are not explicitly considered. The translation of changes in service indicators to monetary values in BCA overcomes the problem of comparability and subjectivity, by converting changes in the different indicators to a common currency that constitutes an objective proxy for impacts on societal welfare. We assert this because economic valuation is based on society's willingness-to-pay, which is a function of citizens' preferences and incomes (which can be considered a measure of the strength of their preferences). Further, it makes explicit the potential gainers and losers from different forestry practices: the general public (now and in the future) as implied by changes in the present value of public-good ES or forest owners as implied by changes in the present value of harvests.

We argue that BCA is desirable as an overarching methodology for evaluating different management practices and the potential need for policy interventions in forestry, because forests also generate public-good ES and society has limited resources for financing the provisioning of public goods (e.g., conservation budgets). To allocate these resources efficiently, unavoidable trade-offs among ecosystem services should be optimized to maximize societal welfare, which is the goal of BCA (Johansson, 1991).

Our case-study indicates that increasing the area of spruce forest managed as continuous cover forestry (CCF) and reducing the area of clear-felling (BAU) would increase the contribution of Swedish production forests to societal welfare as compared to current clear-felling practices, whole-tree harvesting and shorter rotations. Thus our study lends support to the contention that current forestry in Sweden prioritizes wood production over other values (Eggers et al., 2019). Here we show how these values can be derived and included in a welfare-economic evaluation of forestry management practices.

According to economic theory, the marginal benefits of any particular ecosystem service will decline with increases in its own supply (i.e., as implied by a normal down-ward sloping demand curve). Consequently, the desirability of increasing the area of CCF is expected to be diminishing: the greater the area converted, the lower the additional or marginal societal benefits of even greater conversion. At some point, the contribution of forests to welfare will be maximized and additional conversion would reduce welfare, because the opportunity cost of reduced forestry production associated with CCF would be greater than the marginal benefits of additional public-good services. This implies that there exists an optimal trade-off between the area managed with CCF and clear-felling or similar management practices for that matter.

Our BCA also highlights that the balancing of current and future generations' welfare, as reflected in the choice of discount rate, has a decisive impact on the contribution of the different management scenarios to societal welfare. The discount rate plays the key role of determining the best allocation of resources between the present and the future (Gollier, 2011). We show that higher discounting, as generally applied by forest owners, favours wood production over public-good services and hence reduces the desirability of converting to CCF compared to the more intensive management scenarios. On the other hand, greater concern for future generations, as represented by lower rates of discount, favours CCF. Consequently, we identify a conflict in the optimal choice of management practices from society's perspective compared to that of foresters. This illustrates a potential need for policy to coordinate the financing and provisioning of public-good forest ES, particularly for safeguarding the welfare of future generations, if the value of forests is to be maximized for the common good.

Finally, we stress that while CBA provides a powerful tool for environmental management, it should not be, and usually is not, the sole basis for public decisions (Hanley and Spash, 1993); and naturally should be applied carefully. Some argue though that it is entirely inappropriate for informing environmental decision-making (Gómez-Baggethun and Ruiz-Pérez, 2011; Sagoff, 2011), but this we believe would be "throwing the baby out with the bath water" since it can for instance be integrated within participatory processes (Carolus et al., 2018). However, we agree that it is likely inappropriate when ecological thresholds for critical natural capital or irreversible effects on biodiversity are threatened (Farley, 2008). In such cases cost-effectiveness analysis would be the desirable welfare-economic approach, which involves identifying the least costly means to conserve an environmental asset such as an endangered species; thereby making the need for monetary valuation redundant (i.e., when society is in general agreement that the benefits of a proposed conservation project would vastly exceed the conceivable costs).

In summary, our analysis suggests that an increase in the area of CCF (or other environmentally friendly forestry practices) seems motivated from a welfare economic perspective, but how large an area requires further analysis. From this perspective there is, currently, over harvesting of wood to the detriment of public-good forest ES, implying that the area of clear felling is currently exceeding the socially optimal area. Finally, the desirability of increasing the intensity of biomass extraction from forests seems short-sighted due to the negative impacts on public-good ES, particularly on forest carbon storage and the welfare of future generations. Therefore, it is crucial that the value of public-good forest ES is considered in management decisions as a pathway to sustainable forestry. We hope that our approach may provide inspiration for integrating BCA in forestry decision-support systems at multiple scales.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.ecoser.2019.101011.

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