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Environmental impact assessment and monetary ecosystem service valuation of an ecosystem under different future environmental change and management scenarios; a case study of a Scots pine forest

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Abstract

For a sustainable future, we must sustainably manage not only the human/industrial system but also ecosystems. To achieve the latter goal, we need to predict the responses of ecosystems and their provided services to management practices under changing environmental conditions via ecosystem models and use tools to compare the estimated provided services between the different scenarios. However, scientific articles have covered a limited amount of estimated ecosystem services and have used tools to aggregate services that contain a significant amount of subjective aspects and that represent the final result in a non-tangible unit such as 'points'. To resolve these matters, this study quantifies the environmental impact (on human health, natural systems and natural resources) in physical units and uses an ecosystem service valuation based on monetary values (including ecosystem disservices with associated negative monetary values). More specifically, the paper also focuses on the assessment of ecosystem services related to pollutant removal/generation flows, accounting for the inflow of eutrophying nitrogen (N) when assessing the effect of N leached to groundwater. Regarding water use/provisioning, evapotranspiration is alternatively considered a disservice because it implies a loss of (potential) groundwater. These approaches and improvements, relevant to all ecosystems, are demonstrated using a Scots pine stand from 2010 to 2089 for a combination of three environmental change and three management scenarios. The environmental change scenarios considered interannual climate variability trends and included alterations in temperature, precipitation, nitrogen deposition, wind speed, Particulate matter (PM) concentration and CO₂ concentration.

The addressed flows/ecosystem services, including disservices, are as follows: particulate matter removal, freshwater loss, CO₂ sequestration, wood production, NO_x emissions, NH₃ uptake and nitrogen pollution/removal.

The monetary ecosystem service valuation yields a total average estimate of 361-1242 euro ha⁻¹ yr⁻¹. PM_{2.5} (< 2.5 µm) removal is the key service, with a projected value of 622-1172 euro ha⁻¹ yr⁻¹. Concerning environmental impact assessment, with net CO₂ uptake being the most relevant

contributing flow, a loss prevention of 0.014-0.029 healthy life years $\text{ha}^{-1} \text{yr}^{-1}$ is calculated for the respective flows. Both assessment methods favor the use of the least intensive management scenario due to its resulting higher CO_2 sequestration and PM removal, which are the most important services of the considered ones.

Keywords: Forest management, ecosystem services, forest, environmental impact assessment, particulate matter, climate change

Introduction

To ensure an environmentally sustainable future for mankind, we must better manage our human/industrial system by reducing its environmental impact caused by emissions of harmful compounds and resource extractions and by industrial remediation of inflicted environmental damage (e.g., soil sanitation). However, we can also manage ecosystems in a manner such that they also indirectly aid us in achieving these sustainability efforts, e.g., provision more renewable resources, in the best way possible.

In addition to the influence of mankind on ecosystems through its direct actions, e.g., harvest and management practices, changing environmental conditions (caused by mankind and nature) such as climate change, which is important to account for, and their interactions with mentioned management practices also have an influence (FAO, 2012). Future environmental conditions are however not exactly known. In this regard, different environmental change scenarios can be drawn, and some long-term scenarios were specifically depicted by the Intergovernmental Panel on Climate Change (IPCC) (IPCC, 2014, 2000).

To unravel the most sustainable ecosystem management scenario, one should estimate what the sustainability impact will be of the ecosystem under these different scenarios in combination with different environmental change scenarios. Because the endpoint is to predict the effect of human actions in the future, it is worth modelling ecosystems according to different management and environmental change scenarios. Furthermore, the combination with approaches that assess the sustainability (or an aspect of it) is more and more needed. In this study, the focus is on environmental sustainability, considering the effect on mankind and nature (see Figure 1 for an overview).

Figure 1. Overall scheme of the influence of mankind and nature on an ecosystem and the subsequent indirectly induced damage and benefit to themselves. As a case study, we set up a framework to quantify these relationships and effects for a Scots pine forest using different scenarios and methods, shown between brackets.

The main goal of this study is to further the science in the field of environmental sustainability assessment of an ecosystem under different management and environmental change scenarios. This will be exemplified through a case study of a forest, more specifically a Scots pine stand. Different aspects to which improvements are made are addressed in the following subsections: “Environmental sustainability assessment approaches” (section 1.1), “Predicting future environmental change” (section 1.2) and “A case study of a specific forest ecosystem” (section 1.3).

1.1 Environmental sustainability assessment approaches

A particular ecosystem has various interactions with their environment (mankind and the rest of nature), ranging from biomass harvest to recreation, that define their sustainability. The ideal approach to address the environmental sustainability of these interactions results in a single, purely objective (i.e. without needing to involve subjective weighting of different aspects) and tangible (e.g. not using points) outcome. However, such an approach is still non-existent, and we will therefore apply two prominent conceptual approaches.

The first is the ecosystem service approach. As is well described in the Millennium Ecosystem Assessment reports (2005), ecosystem services represent the direct and indirect contributions of ecosystems to humans’ well-being (De Groot et al., 2012). Much research is being performed on the

mapping, assessment and (monetary) valuation of ecosystem services (Brown and Fagerholm, in press; Crossman et al., 2013; Maes et al., 2013; Zulian et al., 2014), developing a quantitative basis for the estimation of ecosystems' value to mankind and thus facilitating policy support and decision-making. We interpret an ecosystem service as a property, function, process or a collection of these factors of an ecosystem that provide a benefit to mankind. Ecosystem services are furthermore subdivided into different categories: provisioning (e.g., food, water), regulating (e.g., removal of pollutants, pollination, etc.), supporting (that support other services, e.g., nutrient cycling, primary production, etc.) and recreational/cultural services (Millennium Ecosystem Assessment, 2005). Other classifications also exist, but, according to De Groot et al. (2010), "we should accept that no final classification can capture the myriad of ways in which ecosystems support human life and contribute to human well-being". Here, we will not focus on ecosystem services that relate to social human well-being. In addition to services, ecosystems also provide disservices, e.g., spreading of infectious diseases, crop damage by pests, emission of volatile organic compounds and allergic reactions to pollen (Dunn, 2010; Escobedo et al., 2011; Lyytimäki and Sipilä, 2009). Because these affect human well-being negatively, they should also be accounted for (Lyytimäki, In press; Lyytimäki and Sipilä, 2009)

One can account for these different aspects of an ecosystem through a set of indicators and then compare their values. To this end, a multi-criteria analysis (MCA) methodology could be eventually used to identify the preferred management scenario (Cinelli et al., 2014; Hails and Ormerod, 2013; Langhans et al., 2014; Wolfslehner and Seidl, 2010). Some studies compare the scenarios using raw indicator outcomes without the use of an additional MCA methodology (Lasch et al., 2010; Temperli et al., 2012). In this case, the selection of the best scenario is generally based on implicit subjective weighting of the indicator values. Such an approach might be reasonable if only a few indicators are considered, but it remains nonetheless strongly based on subjective opinions. One can also make use of principal component analysis to aid judgment, though this does not result in a single outcome (Duncker et al., 2012). To obtain a more broad opinion on a community scale, the opinion/weighting of indicators by stakeholder groups, e.g., environmental organizations, can be included. An additive

utility models can also be used in which stakeholder priority settings are converted into mathematical relationships used to obtain a single numerical outcome, as done by Fürstenau et al. (2007). Similarly, Seidl and Lexer (2013) use a complex framework partially based on a weighting of selected indicators done by some stakeholder groups. A single score can be obtained for both of these methods, but the outcome depends on the subjective priorities/weighting of the stakeholder groups. The authors of the respective studies (Fürstenau et al., 2007; Seidl and Lexer, 2013) conclude that the differences between stakeholder group preferences in fact lead to different outcomes in terms of pinpointing the best management practice. No single (objective) outcome could hence be presented to the reader; no better scenario could be exactly pinpointed. Accordingly, there is a need for methodological development that results in a single or a small set of outcomes based on more/only objective calculations. Furthermore, the aggregated outcomes of the applied multi-criteria analysis methods have no units (they are represented as ‘scores’ or ‘points’), expressing no real tangible quantity and presenting an unclear message regarding the extent of the impact on human well-being of a scenario (choice). If no tangible value is at all attributable to a single MCA outcome, it is difficult to grasp the value of ecosystem services for human well-being, which is relevant in ecosystem service valuation. In this article, the valuation of ecosystem services is therefore put forward through a monetization approach, i.e., valuing ecosystem services in monetary amounts (Baveye et al., 2013; Broekx et al., 2013; de Groot et al., 2012; La Notte et al., 2015; Liekens et al., 2013b; TEEB, 2010). This is not an analysis on financial or economic feasibility/profit, or a cost-benefit analysis of a selected scenario, hence e.g. including management costs, such as that performed by Garcia-Quijano et al. (2005) for climate mitigation through CO₂ uptake. It must be remarked that monetary valuation is not purely objective, though it delivers a tangible overall value and no normalization of services is needed.

Environmental impact assessment can be considered, adding more information on the environmental sustainability of an ecosystem (Figure 1). In this case, the goal is to quantify the impact of a production system on natural resources, human health and ecosystems due to the extraction of resources and the release of pollutant emissions (de Haes et al., 1999; ISO, 2006). This is commonly applied to human/industrial systems, such as product life cycles, while not typically to ecosystems. For example, Garcia-Quijano et al. (2005) use such an approach to assess the land use impact of different

forest management practices. An important aspect of this approach is that it also considers the final impact on nature, whereas ecosystems service assessment focuses only on the eventual benefit to mankind (Figure 1). This approach is even more objective and also presents the outcome in tangible units, though possibly not in a single one. MCA might thus still be needed to obtain a single outcome if more than one output unit is obtained. The advantage of using an environmental impact assessment approach is that it provides a first screening of the environmental sustainability of the system potentially in less and tangible units, facilitating more correct judgment by stakeholders that is later needed for MCA-approaches. –Schaubroeck et al. (2013) noted that, in the environmental impact assessment of an integrated system of forest and wood processing, the forest ecosystem could have the most important share of impact. These authors also used and provided a framework that allows one to better assess environmental impact by including the uptake of harmful compounds (e.g., CO₂) and illustrating the importance of accounting for this uptake. The respective framework is also used in this study. Good management of ecosystems can thus be most important. In this study, we do not include the industrial system, i.e., we do not account for alterations in environmental impact due to a change in the industrial processes associated with a different management approach. However, these impacts do exist, as clearly shown in the case study of Rugani et al. (2015). In our case study, this is most likely negligible, as shown by Schaubroeck et al. (2013), and therefore not accounted for. Note that other approaches exist to aid stakeholders in selecting the most sustainable management scenario: criteria and indicators (Van Cauwenbergh et al., 2007), decision support systems (Gilliams et al., 2005) and knowledge-based systems (Baelemans and Muys, 1998). However, one should make many subjective choices to pursue these, and no overall tangible outcome can be obtained when addressing multiple criteria.

1.1.1 Ecosystem services

An important aspect to address, as illustrated for forests by the literature review (Table S1), is that more relevant services should be considered (see section 2.5) in a more realistic manner and through a sound approach to better pinpoint the best management strategy under quantified changing

environmental conditions (Smith et al., 2013). The most important additional service considered here is particulate matter, responsible for severe damage to human health after inhalation (European Environment Agency, 2013; Kim et al., 2015), removal from the air, a relevant service for a forest ecosystem (Nowak et al., 2014; Schaubroeck et al., 2014) and also possibly for any terrestrial ecosystem, using the modeling framework of Schaubroeck et al. (2014). Moreover, it is important to generally discuss and improve the assessment of pollutant generation/removal as an ecosystem service. In the assessment of such a (dis)service, one should take into account the inflow (or outflow) of pollutants as well. However, the presence of nitrate in ecosystem leachate is currently considered a disservice because of its eutrophication potential (eutrophication indirectly damages ecosystems and human well-being through various pathways, even human health as presented in a report by the World Health Organization (2003)), without taking into account the inflow of eutrophying nitrogen agents (Broekx et al., 2013; Duncker et al., 2012; Liekens et al., 2013b). A solution is presented in this work by considering the ecosystem service equal to the impact of the inflow minus the impact of the outflow, exemplified in nitrogen removal by the respective forest ecosystem. In addition, groundwater recharge is conventionally addressed as a service provided by an ecosystem (Broekx et al., 2013; Duncker et al., 2012; Liekens et al., 2013b). This viewpoint can be questioned, especially because water use will be most likely higher for certain ecosystems, e.g., forests, than for other land uses due to the higher capacity of those ecosystems to evapotranspire (Calder, 2007; Maes et al., 2009). Consequently, it might be more logical to consider freshwater loss as an ecosystem disservice. This is further addressed in section 2.5.

1.2 Predicting environmental change

Environmental change scenarios can be improved. Most studies only focus on the change in CO₂ concentrations and climate-related characteristics, such as temperature and precipitation (see Table S1). However, other environmental parameters are also of high relevance, e.g., nitrogen deposition related to the ecosystem service of nitrogen pollutant removal or generation, as included in the case study. Moreover, another important aspect of predicting environmental conditions is interannual variability. Environmental change scenarios provide long-term predictions of how the climate will

change on average. However, the climatic conditions at each period in time, e.g., for a year, are not exactly predictable because they may still vary due to natural variation, e.g., year 2020 might be colder than year 2019 even though the temperature is predicted to rise on average. Focusing on only weather conditions (precipitation, temperature and irradiation), the uncertainty of near-future predictions is in fact very dependent on this natural interannual variation, see figure 11.8 of the IPCC report (2014). It is therefore definitely important to include an interannual variation because, for example, initial forest growth is sensitive to weather conditions (Cunningham et al., 2006; Dzwonko and Gawroński, 2002; Taeger et al., 2013a, 2013b). In this study, an approach for this is presented, while including alterations in the following environmental change parameters as much as possible: temperature, precipitation, nitrogen deposition, wind speed, particulate matter concentration and CO₂ concentration.

1.3 A case study of a specific forest ecosystem

One of the most relevant terrestrial ecosystem types is the forest ecosystem. Forests covered 31% of the total land area in 2010 (FAO, 2010) and provide valuable goods and services to mankind, such as the provisioning of wood or climate change mitigation, e.g., sequestering carbon dioxide (Pan et al., 2011). We therefore focus on this ecosystem type in this study; however, the conceptual approaches are applicable to any ecosystem type, and the case study results are also interesting for similar research on other vegetative terrestrial ecosystem types. To the best of our knowledge, five different similar studies (including the influence of environmental change and management) on specific forests have been performed to date (Duncker et al., 2012; Fürstenau et al., 2007; Pizzirani et al., 2010; Seidl and Lexer, 2013; Temperli et al., 2012), as comprehensively reported in Table S1 of the Supplementary File.

Many new studies focus on the general quantification/mapping of ecosystem services using very robust, generic models across landscapes, delivering valuable data on ecosystem services on a regional level (Boumans et al., 2015; Brown and Fagerholm, in press; Villa et al., 2014; Zulian et al., 2014). However, there is still a need for thorough detailed studies of ecosystem service supply under different scenarios of environmental change and management, e.g., using high-quality models with the high

data demand, on specific sites because the quality of such (ecosystem service) assessments and valuation of the latter studies can be questioned. Such a detailed assessment is presented in this work.

Materials and methods

2.1 Site description

The forest ecosystem analysed here is a managed Scots pine (*Pinus sylvestris* L.) forest stand, referred to as ‘Scots pine stand’, located in the forest ‘De Inslag’ in Brasschaat, Belgium (51°18’33” N, 4°31’14” E). The original Scots pine stand has been extensively studied. For more information, read the works of Schaubroeck et al. (2013, 2012). The regional climate is temperate maritime with a mean annual temperature of 9.8°C and a mean annual precipitation of 750 mm (Nagy et al., 2006). The soil is loamy sand and moderately humid, with distinct humus and iron B horizons. The groundwater table is usually at a depth of 1.2-1.5 m (Baeyens et al., 1993). A more detailed description of the soil is given by Neiryneck et al. (2002) and Janssens et al. (1999). This forest ecosystem was selected because it is a thoroughly studied stand (Schaubroeck et al., 2013, 2012) for which there is a large amount of data, and because Scots pine is an abundant species in European forests (Skjøth et al., 2008; Tröltzsch et al., 2009).

2.2 Model selection

Models were developed to predict general forest growth. For an overview of the types and models applied in the other studies mentioned in Table S1, see the supporting information section B. Here, we applied the hybrid model ANALysis of FORest Ecosystems model (ANAFORE) that was introduced by Deckmyn et al. (2008) and later updated with a soil sub-model (Deckmyn et al., 2011). The model is discussed more in detail in section B of the supplementary file. Half-hourly, daily or monthly values of temperature, precipitation, radiation, wind speed, humidity and CO₂ and the stand inventory, forest management and soil characteristics are inputs. ANAFORE is a high detailed parameter-rich model and is therefore less suitable if not enough parameter data are available (van Oijen et al., 2013). However, this is not an issue in this case because the model was already applied and validated for the Scots pine stand considered here (Deckmyn et al., 2011, 2008).

Schaubroeck et al. (2014) integrated the assessment of particulate matter (PM) removal on a half-hourly basis into the ANAFORE model and applied it to the Scots pine stand studied here. This particular submodel was also used in this study to assess PM_{2.5} (PM with a diameter < 2.5 µm) and PM_{2.5-10} removal. The input requirements needed to run it are illustrated by Schaubroeck et al. (2014). The parameter values of Schaubroeck et al. (2014) for PM_{2.5} removal by a Scots pine forest are used and are also applied for PM_{2.5-10} removal.

For the PM removal submodel, the calculation of wood area, which is an important variable for the interception of rain, was improved. See section C of the supplementary file for a full description.

To conclude, the ANAFORE model was selected because of the following reasons: (1) it models soil processes (Deckmyn et al., 2011); (2) it is a detailed process-based model (Deckmyn et al., 2008); (3) for a process-based model, it quantifies relatively many ecosystem services and goods, mostly in a detailed manner (Table S1); (4) it has been updated with a model on particulate matter removal (Schaubroeck et al., 2014); and (5) it has been applied and parameterized for the studied site (Deckmyn et al., 2008).

2.3 Management scenarios

The studies described in Table S1 test the management scenarios within a broad spectrum, from no to intensive management, to select the optimal type of management for the forest (Duncker et al., 2012; Fürstenau et al., 2007; Pizzirani et al., 2010) or in light of a specific research question (Seidl and Lexer, 2013; Temperli et al., 2012). Here, the three tested management scenarios are intensive to check whether different outcomes can be obtained by the framework, even at that level of detail, i.e. for quite similar scenarios.

For all scenarios, the considered management of the Scots pine stand is an 80-year rotation period, starting from 10 000 one-year-old trees planted per hectare after a clear felling of the current pine forest in 2010 until the next clear cut in 2089. The initial conditions are those after a virtual clear-cut of the existing 80-year-old forest in 2010. The carbon amounts in the soil are those given by Gielen et al. (2013) and are mentioned in Table S2 of Schaubroeck et al. (2014). The distribution over the soil layers is retrieved from a previous run of ANAFORE on the same site. Two-year-old saplings were

planted. To initialize the soil conditions after a clear cut in 2010, a preparatory model run was performed for a full length of pine rotation. We consider only one tree cohort in the ANAFORE model, which is reasonable because it is a planted forest.

In all scenarios, after 14 years, a cutting occurs in which 30% of the trees are harvested. The subsequent thinning is different between the scenarios. For LOW management, no thinning occurs. For MID and HIGH management, thinning is performed every 5 years, the applied frequency in yield tables for Scots pine in this region (Jansen et al., 1996), starting from year 21. For MID management, half of the wood increment over 5 years is harvested. For HIGH management, all of the increment is harvested. Random trees are cut when thinning. This is a simplification because different thinning procedures exist, e.g., thinning from taller or shorter trees, but a mixture of these procedures is often applied in actual forest practices. Only wood from stems and large branches is harvested when thinning. Roots, needles and small branch residues are left behind in the forest.

2.4 Environmental change scenarios and their parameter values

Three of the discussed studies, shown in Table S1, use different environmental change models to assess different scenarios (Fürsternau et al., 2007; Seidl and Lexer, 2013; Temperli et al., 2012). Two only consider the increase in temperature and precipitation. In addition to these two variables, Fürsternau et al. (2007) also considered an increase in CO₂. Pizzirani et al. (2010) only considered the increase in biotic threats. In this study, monthly values of weather conditions (temperature, precipitation and radiation) are used as model inputs, and yearly values are used for the others.

Simulations were performed from 2010 to 2089 with three different environmental change scenarios that capture the possible trends in environmental changes: one assuming no change as a reference, the current (CUR) scenario, and two alternative future scenarios. The latter two are roughly based on two possible socio-economic incentives and their effect on environmental change. The severe (SEV) scenario is based on an evolution in which the current environmental policy is considered, implying a more economic-growth-oriented vision. The other future scenario, called moderate (MOD), reflects a

development in which more measurements are taken to provide greater socio-environmental sustainability.

Another matter to address when specifying environmental change scenarios is whether there will be a convergence of different communities, i.e., similar climate-related policies among the world-wide communities (IPCC, 2000). Here, we consider a heterogeneous/non-convergent world for both MOD and SEV because we are dealing with more local policies and the implementation of environmental policies still differs among communities, e.g., compare the climate policy of the U.S. with that of Europe (Hayes and Knox-Hayes, 2014). The environmental change scenarios differed in 7 out of the 8 meteorological and environmental variables driving the model: air temperature, precipitation, CO₂ concentration, NO_y deposition, NH_x deposition, wind speed and airborne particulate matter concentration (more specifically that of PM_{2.5} and PM_{2.5-10}). The environmental variable for irradiation was considered not to vary among the scenarios because future irradiation changes are expected to be very minor in Belgium (Campioli et al., 2012). The scenarios are aggregated out of other similar scenarios from different references because no single reference provided values for all considered parameters and, furthermore, we wanted to use site-specific values. For an overview, see Table 1. The environmental parameter values differ on a yearly basis, except for precipitation and temperature, which also differ on a seasonal basis.

Table 1. The aggregated climate scenarios and the scenarios used to model the respective parameters, of which the original names are mentioned (e.g., ‘G+’).

	Moderate (MOD)	Severe (SEV)	Values	Reference
Temperature, precipitation, & wind speed	G+	W+	Table	(van den Hurk et al., 2006)
CO ₂ concentration	A2	B2	Figure 1	(IPCC, 2001)
N deposition (NH _x and NO _y) & PM air concentration (PM _{2.5} & PM _{2.5-10})	Europe	Reference	Figure 1	(Van Steertegem, 2009)
Overall similar IPCC scenario	A2	B2	/	(IPCC, 2000)

The considered changes in temperature, precipitation and wind speed, shown in Table 2, were obtained for the moderate (MOD) and severe (SEV) scenarios for the region of study until 2090 based on the scenarios G+ (equivalent to B family) and W+ (equivalent to A family), respectively, of the

Royal Dutch Meteorological Institute (Demarée, 2008; van den Hurk et al., 2006). On average, a warmer climate with wetter winters and drier summers compared to the current climate was predicted for the moderate and even more for the severe scenario. For precipitation, this is not considered on an average yearly basis because the projected yearly changes are small compared to the internal variability (i.e., smaller than one standard deviation of the estimated internal variability), see figure 11.12 in IPCC (2014), and, specific for Europe, model results do not agree on a yearly corresponding change in precipitation in response to CO₂ increase (IPCC, 2014). Wind speed only influences particulate matter removal and evapotranspiration of water and is forecasted to increase in the G+ (moderate) and W+ (severe) scenarios. All these relative changes in percentages are applied assuming a linear increase over time, e.g., for wind speed increase, a factor of 0.0364 (moderate) and 0.0727 (severe) per year can be derived for the moderate and severe scenarios, respectively.

Table 2. Considered changes for 2100 compared to 1990 in precipitation, temperature and wind speed for the two future scenarios, based on the work of Van den Hurk et al. (2006). The seasons are defined as follows: 'winter' stands for December, January and February and 'summer' stands for June, July and August.

Environmental change scenario	Moderate	Severe
Original name	G+	W+
Temperature		
Global air T (applied to spring and autumn)	+2°C	+4°C
Winter average T	+2.3°C	+4.6°C
Summer average T	+2.8°C	+5.6°C
Precipitation		
Winter average precipitation	+14%	+28%
Summer average precipitation	-19%	-38%
Wind speed		
Average wind speed	+4%	+8%

For CO₂ concentrations, the current scenario concentration was set constant at 390.103 ppmv (IPCC, 2001). Future CO₂ projections for moderate (B2) and severe (A2) scenarios foresaw a gradual CO₂

increase up to 585 and 762.5 ppmv, respectively, in 2090 based on averages of the reference scenario in models ISAM and Bern-CC (IPCC, 2001) (Figure 2).

Future projections of nitrogen deposition and particulate matter (PM) concentrations (Figure 2) are based on a report of the Flemish Environmental Agency (FEA), in which different socio-economic scenarios are applied to predict respective future environmental changes in Flanders, Belgium, until 2030 (Van Steertegem, 2009). For the moderate scenario, the ‘Europa’-scenario was selected, in which environmental change is based on meeting specific European environmental policy directives (Amann et al., 2008) applied to Flanders, which is in line with an increase in socio-environmental sustainability.

For the severe scenario, the ‘reference’ scenario was selected, which is a business-as-usual approach. The FEA-calculated specific future prediction values are valid for the location of the Scots pine stand for the years 2010, 2015, 2020, 2025 and 2030. Linear interpolation was used to obtain values for the years between latter ones, and values after 2030 were considered as remaining constant. These trends in evolution were applied using recalibration based on local measurements/determinations of nitrogen deposition and PM concentrations in the year 2010, illustrated by the following example: new prediction 2015 = (prediction FEA 2015) / (prediction FEA 2010) × (measurement 2010).

The total nitrogen deposition to the soil in 2010 is considered to be 40 kg N ha⁻¹ yr⁻¹ with a share of 0.21 NO_y-N and 0.79 NH_x-N, valid for the period 1992-2007 for the Scots pine stand (Neirynck et al., 2008). The effect of changing vegetation on the dry deposition of PM and thus on the nitrogen deposition is not considered because this was only responsible for 20% of the total nitrogen deposition (Neirynck et al., 2007). The FEA and IRCEL, the Belgian Interregional Environment Agency, provided a yearly concentration for 2010 of 24.55 µg m⁻³ PM_{2.5-10} and 16.77 µg m⁻³ PM_{2.5} (with a resolution of 3*3 km), for which the methodology of obtainment is explained by Schaubroeck et al. (2014).

To model PM removal, the half-hourly precipitation and PM concentration need to be known (Schaubroeck et al., 2014). For 2010, these values were measured for precipitation and were calculated

by IRCEL and FEA for PM as addressed above. Half-hourly precipitation and hourly PM values for other years were obtained via recalibration using the yearly values, e.g., half-hourly precipitation values of 2011 = (half-hourly precipitation values of 2010) $\times$ (yearly precipitation 2011) / (yearly precipitation 2010). This is not calculated using monthly values for precipitation instead of yearly values because unrealistic results could be obtained due to greater variation between months than between years. Note that every year has similar rain and PM patterns as the reference year 2010. Humidity is set constant at a high value of 0.7 because the climate of the Scots pine stand is humid.

Figure 1. Nitrogen (N) deposition, CO₂ and airborne particulate matter (PM) values over time for the current (CUR; blue), moderate (MOD; green) and severe (SEV; red) environmental change scenarios.

2.4.1 Interannual variability of weather conditions

Only interannual climate variability of future climate uncertainty was included. This was performed practically via a new approach, exemplified by the case study. If one assumes that the current scenario is close to that of the previous 10 years (1999-2008), 80 random year samples (with monthly radiation, temperature and precipitation) may be taken out of this pool to obtain one random period of 80 years as the weather input for a run from 2010 to 2089. For example, for one sample, the weather conditions of year 2011 are similar to those of year 2000, and those of year 2012 are similar to those of year 2006, etc. The weather values for the period 1999-2008 were obtained from local measurements (Kint et al., 2012). Fifty random periods were thus created to serve as the weather input for the model runs of the current scenario. Taken together, this created a natural variation in weather conditions. The annual environmental changes in temperature and precipitation (Table 2) per scenario were superseded to obtain the weather inputs for the two future scenarios.

2.5 Ecosystem services and their monetary valuation

Ecosystem services are variable over time and space (Lyytimäki and Sipilä, 2009). For example, a forest may reduce runoff to a nearby river, lowering the risk of flooding, but this could lead to a water shortage later on in another region that stores water from this river in a reservoir. Assessments of these services may thus be very case-specific and should therefore be considered for fixed time and space boundaries in practice if applicable.

Only ecosystem services provided during the forest management period were accounted for. The area benefiting from the services varied between the services, from local (water recharge) to global (global warming potential). We first selected ecosystem (dis)services and the processes or aspects responsible for them, which could be directly attributed to the ecosystem. A service should, after all, be the specific result of a function or activity of the forest. In practice, for a regulating service in pollution remediation, these are processes that lead to the enhanced or active uptake of polluting compounds and/or the processing of them to forms that are not/less harmful. Beyond this, our selection of ecosystem (dis)services was also restricted to those for which monetary values were available. In addition to this criterion, data should be available or modeled, by ANAFORE in our case, to account for a (dis)service. Table 3 lists the ecosystem (dis)services considered in this study.

In the literature reviewed in Table S1, wood production, because it has long been considered the primary function of forests, and biodiversity are accounted for by all 5 studies in a certain manner. Carbon sequestration is considered by three of the studies and groundwater recharge by two. In our study, we considered the following services: wood production, carbon sequestration, water evapotranspiration, $PM_{2.5}$ and $PM_{2.5-10}$ removal, NO_x emissions, NH_3 processing and nitrogen pollution/removal. Biodiversity of the forest itself and alterations were not assessed according to the following reasons: (1) forest biodiversity and the differences among scenarios will be small for a planted and intensively managed forest and (2) there is a lack of knowledge with which to value/monetize its benefit towards human well-being (Cardinale et al., 2012). See section J of the supplementary material on the current debate about the consideration of biodiversity as an ecosystem service. Stored nutrient amounts and cultural/recreational services were also not considered in the present ecosystem service valuation. Accordingly, the total amount of nutrients already stored in an

ecosystem is considered by some as an ecosystem service, e.g., the carbon stock (Broekx, 2013; Duncker et al., 2012; Ninan and Inoue, 2013). However, if services provided by an ecosystem over a period of time are quantified, only the increase or depletion during that time period and the maintenance of the stock should be accounted for. The previously stored nutrient amounts, such as that of carbon and nitrogen, are the result of sequestration before the studied time period, which is thus outside of the system boundaries. Finally, recreational/cultural services were not part of environmental sustainability, and there are several issues in their quantification to substantiate their exclusion from this study (see further argumentation in section J of the supplementary file).

To value ecosystem services, monetary values may be attributed to them (Baveye et al., 2013; de Groot et al., 2012). Specific to the region of Flanders, such values have been developed by the Flemish Institute for Technological Research (Broekx et al., 2013; Liekens et al., 2013b) and are used herein. Economic values can be attributed to ecosystem services via different approaches (La Notte et al., 2015). For provisioning services, this is possibly their straightforward normal market price, if it is a merchantable product. Contingent valuation or choice experiments using public surveys, leading to willingness- to-pay for a service, represent other types of approaches, typically used for recreational/cultural services (Liekens et al., 2013a). For regulating services, a first calculation option is the avoided damage cost, and the second option is the avoided abatement cost. Note that the variety of methods induces different outcomes for a certain service and thus variability in its monetization (Kumar et al., 2013). Consequently, it is often infeasible to use a single approach consistently, leading to inevitable uncertainty in the outcome. In fact, the selection of the monetization approach will induce just a share of the associated uncertainty, as a lot of it comes into play for the monetary valuation of ecosystem services as thoroughly discussed by Boithias *et al.* (2016). Other methods to monetize ecosystem services are not addressed here. For disservices, the negative value of the opposite service is considered. Negative monetary values should be regarded as the cost mankind must pay to negate this disservice. Table 3 shows the monetization approaches used for the different (dis)services considered. Because monetization of ecosystem services specific for Flanders is already performed by mentioned authors (Table 3), their presented values, and the selection of monetization approaches have been mainly used as such. We agree with their arguments regarding why certain monetization

approaches have been selected for the respective (dis)services. Note that these authors were often restricted in their selection by the available amount of data and information. For some specific services, we had to select the approach and associated monetary values ourselves. For wood production, we selected market pricing, as is commonly done (Broekx et al., 2013; Duncker et al., 2012; Fürstenau et al., 2007; Pizzirani et al., 2010) and is explained later. Regarding air pollutants, we were consistent and selected the ‘avoided damage cost’ approach as applied for particulate matter and argued by Liekens et al. (2013b). Because monetary values may vary from year to year, 2010 was chosen as the reference year because it is the first year of the management period.

Table 3. Ecosystem (dis)services considered for the forest ecosystem with their monetary valuation, and the characterization factors for environmental impact assessment based on values from ReCiPe version 1.08 (2009), Alvarenga et al. (2013) and Pfister et al. (2009). Calculation of the service is performed via modeling using ANAFORE (Deckmyn et al., 2011, 2008) or via data retrieval. Negative monetary values are attributed to disservices in the strict sense. Nitrogen (N)-removal for water purification may also be a disservice if there is a depletion of the soil N stock. DALY: disability-adjusted life years; PM: particulate matter; PO: photochemical oxidant.

Monetary valuation of ecosystem (dis)services						Environmental impact assessment		
Ecosystem (dis)services		Calculation		Monetary valuation		/		
Ecosystem (dis)service (type)	Description	Source	(Additional) calculation	Value(s)	Type	Source	Impact categories	Characterization factors
Production of wood (provisioning)	The amount and quality of stem wood produced and harvested	Modeled		Price of standing stem wood (euro m ⁻³) for different circumferences	product price	Experts forestiers	/	
Sequestration of CO ₂ (regulating)	Quantity of CO ₂ stored as carbon in the forest	Modeled		20 euro ton ⁻¹ CO ₂	Avoided abatement cost	(Aertsens et al., 2013)	Global warming	1.4E-06 DALY kg ⁻¹ CO ₂ 7.93E-09 species*yr kg ⁻¹ CO ₂
Processing of NH ₃ (regulating)	Processing of gaseous NH ₃ after uptake from air	Data	51.44% of NH _x -N deposition (Neiryneck et al., 2007)	30 euro kg ⁻¹ NH ₃	Avoided damage cost	(De Nocker et al., 2010)	Marine eutrophication	Dissolved in water: 1 kg N eq. kg ⁻¹ N gaseous or particulate 0.092 kg N eq. kg ⁻¹ NH ₃ 0.039 kg N eq. kg ⁻¹ NO _x 0.087 kg N eq. kg ⁻¹ NH ₄ ⁺ 0.028 kg N eq. kg ⁻¹ NO ₃ ⁻
Emission of NO _x (disservice)	Emission of NO _x to the air	Data	5.29% of N deposition (Neiryneck et al., 2007)	0.6 euro kg ⁻¹ NO _x *	Avoided damage cost	(De Nocker et al., 2010)		
Water purification/pollution via N-removal/emission (regulating)	The net amount of eutrophication potential (kg N eq.) of the forest (see section 2.5)	Data & Modeled	kg N eq. input – kg N eq. output (based on the values of marine eutrophication)	5 euro kg ⁻¹ N**	Avoided abatement cost	(Broekx et al., 2013)	Terrestrial acidification PM formation	1.42E-8 species*yr kg ⁻¹ NH ₃ 3.25E-9 species*yr kg ⁻¹ NO _x 8.32E-5 DALY kg ⁻¹ NH ₃

		5.72E-5 DALY kg ⁻¹
		NO _x
PO formation		3.9E-8 DALY kg ⁻¹ NO _x

Continuation Table 3

Monetary valuation of ecosystem services						Impact assessment		
Ecosystem services		Calculation		Monetary valuation		/		
Ecosystem service (type)	Description	Source	(Additional) calculation	Value(s)	Type	Source	Impact categories	Characterization factors
Enhanced removal of particulate matter (PM) (regulating)	The amount of PM present in air that is taken up by the foliage and ends up in the soil	Modelled	See chapter 3	150 euro kg ⁻¹ PM _{2.5} ; 25 euro kg ⁻¹ PM _{2.5-10}	Avoided damage cost	(De Nocker et al., 2010; Liekens et al., 2013b)	PM formation	2.6E-04 DALY kg ⁻¹ PM
Loss of freshwater (dis-service)		Modelled	rain – infiltration = evapotranspiration + runoff [#]	-0.075 euro m ⁻³ H ₂ O Tax for water extraction	product price	(Broekx, 2013)	Freshwater consumption	0 DALY m ⁻³ -2.52E-9 species*yr m ⁻³
/							Resource use	278 GJex ha ⁻¹ yr ⁻¹
/							Land occ.	1.2E-4 species*yr ha ⁻¹ yr ⁻¹

*: This service includes the indirect effect on the ozone level.

**: Broekx et al. (2013) give a low, 5 euros, or high, 74 euros, kg⁻¹ N removed from water. Expert knowledge of Prof. Dr. Ir. Siegfried Vlaeminck indicates 5 euros as the correct value.

[#]: runoff is negligible for the Scots pine stand because it has almost no slope.

We elaborate more on the considered services in the following text. Furthermore, we explain why some considered services or approaches presented in the literature are not taken into account in our study. The **provisioning of freshwater** through seepage is accounted for in some studies as a service provided by forests (Fürstenau et al., 2007; Ninan and Inoue, 2013). This could be questioned because it is the complete hydrological cycle that produces rain falling on land that may end up as available freshwater. Attributing this service solely to a terrestrial ecosystem, such as a forest, is not correct. It is not the case that, without the ecosystem, there would be no groundwater recharge. The land ecosystem may, however, influence the fate of freshwater through its influence on runoff, evapotranspiration and infiltration and could thus locally/regionally influence the available stock of freshwater. Runoff and evapotranspiration may in fact lead to a potential loss of freshwater locally because there is less decrease in local groundwater reservoirs. Loss due to evapotranspiration has been noted previously (Jobbágy and Jackson, 2004; Maes et al., 2009). Note that if runoff ends up in another natural freshwater reservoir, it may not be lost. Additionally, on a larger scale, evaporated water could return again as freshwater at another location (Keys et al., 2012). In the studied Scots pine forest, the landscape is flat, and the soils are permeable; surface runoff is thus not significant. It is also situated in an area where a relevant share of infiltrated water later ends up as freshwater through human/industrial groundwater extraction (Broekx, 2013). Though water scarcity is not an urgent issue in the region, it might be in the near future due to depletion of groundwater reserves (note that Flanders is importing approximately 20% of its drinking water), as recently reported by the Flemish government (Rekenhof aan het Vlaams Parlement, 2014). Additionally, if the groundwater table is high enough, tree roots may be able to directly take up ground water, and thus potential freshwater, in addition to rain percolating through the soil (Dawson, 1996; Jobbágy and Jackson, 2004). This was clearly demonstrated by a study on another Scots pine stand with sandy soil in the same region (Belgian Campine), in which the water table contributed, at a certain point, up to 98.5% of the water uptake by vegetation (Vincke and Thiry, 2008). If we only consider the local benefit, evapotranspiration could be regarded as a regulating ecosystem disservice, acting only as a loss of freshwater. This will be accounted for in this study. Duncker et al. (2012) only considered runoff and neglected

evapotranspiration. The prevention of erosion and reduction of the impact of flooding through water retention (Stürck et al., 2014) by ecosystem services are not relevant because the Scots pine stand is not located in an area where this is important (Broekx, 2013). However, evapotranspiration has a cooling effect on the surface, counteracting the temperature increase induced by an increase in greenhouse gases (Bonan, 2008). It is, however, difficult to quantify the monetary value of the ecosystem service provided through this cooling effect, and it is therefore not considered here. Moreover, evapotranspiration acts as a supporting service for ecosystem functioning and thus other services (Maes et al., 2009; Muys et al., 2011). To account for all the supported services provided through evapotranspiration is yet again a hard nut to crack and consequently not done here. Overall, consideration of freshwater-related services is site-specific. Our approach can still be questioned, though it is reasoned why it is favored for this site, because further research is needed to take into account all (aspects of) water-related ecosystem services (Muys et al., 2014).

Water may not be provided directly by an ecosystem, though its composition may be altered. **(Water) purification** is an important ecosystem service that has been put forward many times (Duncker et al., 2012; Ninan and Inoue, 2013). Specifically, there is a water input in the ecosystem with a certain pollutant content, e.g., nitrate, and, after leaving the system, its content may be reduced/the water quality may be improved. In this paper, only nitrogen content is considered under the heading of water quality. To only account for the quality of the water leaving the system is incomplete, because the occurrence of pollutants already present in the initial input, rainfall and deposition is not considered. Broekx et al. (2013) and Liekens (2013b) do however only consider the amount of nitrate-nitrogen leaving the system as a disservice. In fact, the forest ecosystem must cope with the total nitrogen input from rainfall and dry deposition, and the service provided is the amount that does not end up in the water, increasing the water quality. This service is provided by the ecosystem via the uptake of input nitrogen into biomass, converting it into primarily non-harmful dinitrogen gas via microbial processes. A disservice may be the extra presence of N in water flows through depletion of the nitrogen stocks.

In addition, the damaging effect depends not only on the amount of nitrogen but also on the forms, e.g., nitrate, in which this amount is present. By only focusing on the eutrophication potential, which is

relevant when considering water purification, of the nitrogen compounds, we may convert all flows to kg N equivalents according to their midpoint eutrophication potential using the values of an impact assessment methodology (given in Table 3) and subsequently sum them. For nitrogen compounds in water flows, this factor is simply the amount of nitrogen per compound, e.g., $0.78 \text{ kg N kg}^{-1} \text{ NH}_4^+$. For nitrogen in gaseous fluxes and particulate matter deposition, these values are lower due to their lower potential to end up in water flows (Goedkoop et al., 2009). Because the uptake of these fluxes is influenced by the forest, this is more suitable. Ultimately, through enhanced dry deposition via plant surfaces, the forest may aid in bringing N compounds from air into water and thus actively contribute to eutrophication. After obtaining the single summed value in kg N equivalents, we may convert it to a monetary amount by multiplying by 5 euro kg N^{-1} , as mentioned in Table 3. To accurately calculate this value, the composition of nitrogen compounds must be known. For each environmental change scenario, the total amount of $\text{NH}_x\text{-N}$ and $\text{NO}_y\text{-N}$ inputs are already given (see section 4 of Materials and Methods). Based on the values of Neirynck et al. (2007), specific component amounts can be obtained. $\text{NH}_x\text{-N}$ consists of dry-deposited $\text{NH}_3\text{-N}$ (51.44%) and $\text{NH}_4^+\text{-N}$ (17.06%) and wet-deposited $\text{NH}_4^+\text{-N}$ (31.50%). $\text{NO}_y\text{-N}$ consists of wet-deposited $\text{NO}_3^-\text{-N}$ (42.64%) and dry-deposited $\text{NO}_3^-\text{-N}$ (25.58%), $\text{HNO}_2\text{-N}$ (21.71%) and $\text{HNO}_3\text{-N}$ (10.08%). The dry-deposited amounts of $\text{HNO}_2\text{-N}$ and $\text{HNO}_3\text{-N}$ are considered to be $\text{NO}_3^-\text{-N}$ amounts, as in the work of Schaubroeck et al. (2013). The nitrogen leaving the system is considered to be 100% NO_3^- via drainage and there is also the gaseous emission of NO_x , 5.29% of the N deposition (Neirynck et al., 2007).

Two services regarding air purification or pollution by nitrogen compounds are also considered: the net **emission of NO_x** and the uptake with subsequent net **processing of NH_3** ; this is based on the values given in previous paragraph. These flows are net flows, implying that both input and output are indirectly accounted for.

Through harvest, wood is provided to mankind. The price for the ecosystem service of **wood provisioning** is that of the market price per cubic meter of standing wood (€ m^{-3}) prior to harvesting as a function of its circumference (cm) at 1.5 m. These values are obtained from the Belgian federation of forestry experts (“Fédération Nationale des Experts Forestiers,” 2013). Price data for Scots pines from

the year 2010 are used here. See section D of the supplementary file for these values and further discussion.

2.6 Environmental impact assessment (methodologies)

To assess the environmental impact of the forest ecosystem through resources and emissions, the framework of Schaubroeck et al. (2013) is applied, and an overview of impact methods and values is presented in table 3. In this framework, the absorption of harmful compounds, e.g., CO₂, by the forests is a negatively valued impact, reflecting the effect of remediation. The impact on three areas of protection is considered: natural systems/ecosystems, human health and natural resources (de Haes et al., 1999). Similarly, the ReCiPe 1.08 method (Goedkoop et al., 2009) was selected to assess the impact of emissions and land uses on ecosystems, expressed as diversity loss (species*yr), and human health, expressed as disability-adjusted life years (DALY), i.e. number of years of healthy life lost. The characterization of the effects are explained in the ReCiPe report (Goedkoop et al., 2009). For example, the effect of global warming, thus temperature increase induced by greenhouse gas emissions, on human health is assessed through an increase in malnutrition (via decrease in agricultural production), malaria and diarrhea (both diseases which occur more if temperature rises), cardiovascular diseases (linked to an increase in number of days with extreme temperature), coastal and inland flooding (due to increase in precipitation and sea level rise). These are not all the indirect effects of global warming on human health and are quite rough estimations. Regarding the impact category of marine eutrophication, eutrophication in coastal and marine waters where a share of the freshwater and its associated nitrogen compounds end up, no quantification of endpoint diversity loss is yet available, though this effect is acknowledged (Goedkoop et al., 2009). Hence, this is simply expressed as kg N equivalents. Furthermore, the impact of the net loss of freshwater, mainly through evapotranspiration, on human health and ecosystem/species diversity is also assessed via the methodology of Pfister et al. (2011, 2009) in which the local and marginal impacts of the consumption of freshwater of lakes, rivers or aquifers is assessed. For this aspect, specific values for Belgium were used and are given in Table 3. For the impact on human health, the value is 0 DALY m⁻³ because the methodology of Pfister et al. (2009) for developed countries such as Belgium assumes that the loss of

freshwater is easily dealt with (by purchasing it from other countries), which can be questioned given our discussion in section 2.5.). The only resource deprived from nature by the intensively managed hence non-natural forest ecosystem is the growth of natural vegetation that could not grow during the time of occupation. This is assessed as the deprivation of natural net primary production (NPP), expressed in exergy and modeled via a global vegetation model (Haberl et al., 2007), using the cumulative exergy extraction from the natural environment (CEENE) methodology (Alvarenga et al., 2013; Dewulf et al., 2007) based on the same reasoning of Schaubroeck et al. (2013). The CEENE characterization factor for land occupation at the exact location (defined by its coordinates) of the Scots pine stand is $278 \text{ GJ}_{\text{ex}} \text{ ha}^{-1} \text{ yr}^{-1}$ (Alvarenga et al., 2013). Because the natural NPP production is induced by the combination of biotic and abiotic conditions (such as rain and sunlight), these are also indirectly accounted for as resources.

The outcome of the overall environmental impact assessment is represented by a set of four values that represent damage to human health (DALY), ecosystems (species*year), marine eutrophication (kg N equivalents) and resources (CEENE). Because a single value was not obtained and cannot be aggregated in a scientifically sound manner (Schaubroeck et al., 2015a), the scores from these indicators need to be interpreted together.

Results & discussion

3.1 Case study

The difference in gross forest flows, such as carbon dioxide sequestration, is not the primary focus of this article, but interesting findings can still be drawn from them. This is discussed in section E of the supplementary file. The presented modeled values deviate considerably, which illustrates the influence of the natural variation of weather effects, and may thus be considerable in terms of ecosystem growth and its delivered services overall. Additionally, our result emphasizes the relevance of considering the right initial soil conditions (here just after a clear-cut) and using a forest growth model that includes a soil module (Deckmyn et al., 2011), which was not used in the other studies mentioned in Table S1 except that of Seidl and Lexer (2013).

3.1.1 Monetary valuation of ecosystem services

The interpretation and comparison of monetized ecosystem services is more straightforward because these services are all presented in one tangible unit. The profile over time of the provisioning of services is highly similar for all 9 scenario combinations, one of which is presented in Figure 3. A total positive monetary balance is obtained only after 4-5 years, with a cumulative positive balance over 10-11 years, i.e., only then the Scots pine stand is estimated to provide a beneficial overall service to mankind, according to our results. This is due to the high CO₂ losses and nitrate leaching in the early years, explained in section E of the supplementary file. After this period, almost all services maintain a linear increase except CO₂ sequestration, which decreases in slope, and wood harvest, which occurs in steps associated with harvest operations over time. Clear-cutting at the end of the management period is responsible for an important share of the monetary value of this service, leading to the steep increase at the end.

Figure 3. Ecosystem services provided by the Scots pine stand for the moderate environmental change scenario with MID management, presented in monetary values and cumulative over time. The standard deviation induced by the natural variation in weather conditions is represented by grey shading for the total.

In Figure 4 (bottom), the results are presented for the 9 different combinations of scenarios, ranging from 361-1242 euro ha⁻¹ yr⁻¹. By far, the most important service in terms of monetary value is PM_{2.5} removal, with 622-1172 euro ha⁻¹ yr⁻¹. When not considering this service, the total balance would be negative for all scenarios. Next in line is CO₂ sequestration (168 – 371 euro ha⁻¹ yr⁻¹). PM_{2.5-10} removal, NH₃ removal, and wood production all have a yearly average of approximately 100-200 euro ha⁻¹ yr⁻¹. The largest disservice is the loss in freshwater through evapotranspiration by the Scots pine stand, circa 440 euro ha⁻¹ yr⁻¹. Regarding nitrogen, there is a net pollution, meriting 220 euro ha⁻¹ yr⁻¹. However, these last two services essentially do not vary between the different scenarios. The emission of NO_x is a negligible disservice, with a value lower than 5 euro ha⁻¹ yr⁻¹.

The differences between the environmental change scenarios (60 euro ha⁻¹ yr⁻¹ less for the severe and 289 euro ha⁻¹ yr⁻¹ less for the moderate scenarios, on average, than the 939 euro ha⁻¹ yr⁻¹ for the current scenario) are not large and can be mainly attributed to the discrepancy in PM removal and to a lesser degree to NH₃ removal (and to the other services to a small extent); both of these discrepancies can be allocated to the differences between the lower input/airborne concentrations of these pollutants in the future scenarios compared to the current one. Essentially, less ‘pollution’ corresponds to a lower ability of a forest to remove pollutants and to a lower provisioning of the respective services. Note, however, that nitrogen also serves as a nutrient, and a complete depletion in the nitrogen input can be detrimental.

Concerning management scenarios, CO₂ sequestration, PM removal (induced by LAI differences, as discussed in section E of the supplementary file) and wood provisioning are most differentiated and thus largely responsible for the differences between these scenarios. Overall, compared to the MID management scenario, the LOW scenario is estimated to have a 1.25-1.30 times higher total monetary value and the high scenario a 1.71-1.92 times lower value of ecosystem services, favoring the lowest amounts of thinning. The difference in CO₂ sequestration is forecasted to be the largest, relatively, and PM removal and wood provisioning are similar in their relative differences. Selection of the management scenario is estimated here to have (and can have) a considerable impact on the services delivered by a forest. If one only regards the provisioning of wood, the same trend is only visible to a

much lesser extent (see Figure S6 of the supplementary file). Regarding all services, the LOW management scenario is preferred. Certain tradeoffs and synergies might exist between the services, which are discussed in section G of the supplementary file.

Figure 4. Comparison of the results of the 9 different combinations of management (low, mid and high) and the three different environmental change scenarios: current (CUR), moderate (MOD) and severe (SEV). Yearly averages of the environmental impact assessment and monetary valuation of ecosystem services are given. The environmental impact assessment expresses the impact on human health as disability-adjusted life years (DALY) and the impact on ecosystems as ecosystem/species diversity loss; that of resource consumption is constant at $278 \text{ GJex ha}^{-1} \text{yr}^{-1}$ and the difference in eutrophication impact is only discussed in the text. The standard deviation induced by the natural variability of weather conditions (precipitation, irradiation and temperature) (see section 2.4) is depicted with error bars for the total values.

3.1.2 Environmental impact assessment

The environmental impact assessment is expressed in four units: disability-adjusted life years (DALY), species diversity loss, kg N equivalents (marine eutrophication) and cumulative exergy

extraction from the natural environment (CEENE). DALY and species diversity loss are shown in Figure 4. Because CEENE is only quantified per hectare, $278 \text{ GJex ha}^{-1} \text{ yr}^{-1}$, there is no difference in this unit between the scenarios. Concerning the impact on human health, a clear positive effect is estimated in all cases, $0.014\text{-}0.029 \text{ DALY ha}^{-1} \text{ yr}^{-1}$ equal to $5.0\text{-}10.6 \text{ days ha}^{-1} \text{ yr}^{-1}$ prevented by the forest ecosystem. Over a complete management cycle, this represents a prevention of $1.1\text{-}2.3$ disability-adjusted life years ha^{-1} . The largest contributor is the uptake of CO_2 of at least 85%. The rest is attributed to PM removal. Concerning the impact on the biodiversity of natural systems by the considered forest, there is an estimated loss in biodiversity mainly due to the intensive management of the forest. The predicted loss is $1.2 \text{ E-}04 \text{ species*yr ha}^{-1} \text{ yr}^{-1}$, which is the diversity loss induced by the non-natural nature of the planted Scots pine stand: the general difference in biodiversity between a natural system and an intensive forest, such as the Scots pine stand, in Europe (Goedkoop et al., 2009). As already reasoned in section 2.6, the discrepancy in biodiversity among the scenarios will be low and, therefore, the same biodiversity loss is assumed for each scenario. Additionally, freshwater loss likely leads to a minor loss of diversity. CO_2 uptake partially counteracts this biodiversity loss (with 46-101%) by preventing diversity loss on a large scale, normally induced by atmospheric CO_2 . Regarding (marine) eutrophication, also leading to a projected diversity loss but only expressed in kg N, the range is very narrow between the scenarios: $43.2\text{-}47.2 \text{ kg N ha}^{-1} \text{ yr}^{-1}$, with the LOW scenario having the lowest values. However, this shows that the forest would have a negative impact on its environment concerning (marine) eutrophication because the forest aids in bringing airborne particulate nitrogen into the leached freshwater via dry deposition. However, the quantified diversity loss does not completely cover the damage to the ecosystem (quality) because some impact categories are not yet expressed in terms of diversity loss, such as marine eutrophication.

Here, CO_2 sequestration and PM removal are the most important fluxes and also differ considerably between management and, to a lesser extent, between environmental change scenarios. For each aspect, the LOW management scenario comes out on top in this assessment approach.

3.1.3 Implications for the case study

In our case study, the LOW management scenario is predicted to be the best according to both approaches. However, this is only based on a selected amount of ecosystem services/flows and not all of the relevant ones (see section 2.5). Therefore, the total values do not represent the total impact nor the value (of all services) of the forest. First, forest trees can deliver different biomass products, e.g., wood, roots. When only considering wood as a product, the impact and the provided services of the forest can be fully allocated to the wood produced, with repercussions on the environmental impact profile. This is further discussed in section H of the supplementary file.

The specific Scots pine stand had an approximate selling price of 16000 euro ha⁻¹ in 2010 (price retrieved from the owner, the Agency for Nature and Forestry). Considering the same ratio of rental and selling price as in the nearby city of Brasschaat yields a rental price of 143.6 euro ha⁻¹ yr⁻¹. This is approximately a factor 2.5-8.6 lower than the value calculated here that is provided to mankind by the forest through ecosystem services, illustrating the current estimated undervaluation of these services by society.

3.2 Discussion of methodological approaches

The differences in values between the monetary ecosystem service valuation and the environmental impact assessment highlight the distinctions between these methods and also acknowledge that results may vary according to the applied environmental sustainability assessment method. The most important differences are as follows: PM removal is by far the most important for ecosystem service valuation, but CO₂ sequestration is most important for environmental impact assessment; biomass (wood) production is not accounted for in the latter case, and biodiversity loss is not considered an ecosystem service. Straightforward comparison of the different outcomes is difficult as these are different types of approaches. A possible reason for the higher importance of PM removal for the monetary valuation of ecosystem services is that monetization of this service is specifically developed for Flanders, a very densely populated area within Europe and thus more harm is caused by PM. Human health damage by PM in ReCiPe is characterized on a European level. Moreover, ReCiPe does not distinguish between the difference in health damage between PM_{2.5} and PM_{2.5-10}, even though research pinpoints that this is the case (De Nocker et al., 2010); however, Perrone et al. (2013) argue

this matter. Furthermore, the endpoint for ecosystem damage is assessed as species diversity/richness, which directly covers only one aspect, information. As discussed in section 5 of the Materials and Methods, the value of biodiversity is not yet quantified. Theoretically, an increase in richness does not ensure a change in ecosystem processes (de Souza et al., 2013). An improvement would be to assess the functional diversity (de Souza et al., 2013). In the ReCiPe methodological guidelines, however, authors have already noted that the damage to ecosystems also addresses the disruption of mass and energy fluxes, in addition to information (Goedkoop et al., 2009). Interestingly, other authors have explored this issue with approaches based on changes in mass and energy or in exergy fluxes, storage and dissipation (Maes et al., 2011; Schaubroeck et al., 2012; Silow and Mokry, 2010).

However, ecosystem services and environmental impact assessment clearly overlap (see, for example, Table 3) and are increasingly integrated. For example, the uptake of harmful compounds considered by the framework of Schaubroeck et al. (2013) and used in this study is based on regulating ecosystem services. In fact, in that framework, more than just an ecosystem service is accounted for because the beneficial effect on the ecosystem is assessed in addition to that on human health. In our study, this was accounted for by using the midpoint characterization of marine eutrophication to convert all N-flows into kg N equivalents in the ecosystem service valuation. Novel environmental impact assessment methodologies have been developed to assess the potential damage on (the provisioning of) ecosystem services (Arbault et al., 2014), mostly for different land uses (Koellner and Geyer, 2013; Saad et al., 2013). However, an essential issue should not be forgotten: the conceptual difference between ecosystem services and environmental impact assessment is that the first only considers the final benefit to mankind but the latter accounts for the total environmental impact. Note that ecosystem services can also be expressed using environmental impact assessment characterization factors and that there is also a potential to express environmental impact assessment results in monetary values (Pizzol et al., 2015). A solution to resolve this limitation of ecosystem services assessment is to redefine its concept to ultimately consider the well-being of both humans and nature. However, this limitation might be irrelevant because Schaubroeck (2014) and Schaubroeck et al. (2015b) argue that sustainability is an anthropocentric concept, making humans' well-being the only

finality; nevertheless, the indirect impact on mankind via an effect on nature should be accounted for in sustainability assessment. Considering that this is also an ethical issue, the development of a clear objective universal approach is not straightforward.

3.3 Monetization of ecosystem services, the holy grail?

In this study, monetary valuation of ecosystem services was performed. Although it allows placing a single value on all services provided, an economic value does not (completely) represent/capture the real value, the benefit to mankind, and the given value depends on the socio-economic framework used to obtain it, e.g., avoided damage, as discussed by La Notte et al. (2015). Baveye et al. (2013) reviewed different scientific opinions on this matter. Consequently, there is still a need to express ecosystem services in a unit that better represents their real value (Baveye et al., 2013). However, given the economics-oriented society in which we live, monetization makes the concept more tangible (Quine et al., 2013). It is better to simply compare different alternative scenarios that influence the supply of ecosystem services (Kumar et al., 2013), as in the present study. Today, monetization is necessary to easily account for this in our society. Using these values, one may readily consider ecosystem services as economic products. As a step further, one could thus financially reimburse landowners for the services provided by their land through payment for ecosystem services (PES). If we additionally consider these services as tradable entities without fixed prices, different market mechanisms are set loose on the prices that alter them over time. However, it is not guaranteed that these changes in economic price represent changes in the fundamental value of the service to mankind, such as the volatility of the carbon emissions market (Chevallier, 2011). Governments or institutions should control, potentially even fix, these prices. Inversely, this might induce rent-seeking. In this particular accounting/policy method of fixing prices, a difference should be discerned between services that improve the quality of life of the total (global) community, such as climate change, or that of local/regional communities (Kumar et al., 2013). Several of the potential downfalls induced by PES and the necessary regulation for their restriction are discussed in detail by Kronenberg and Hubacek (2013).

In addition, issues related to economic mechanisms such as discounting remain, as exemplified in this paragraph. Consider fixed prices and that the services are provided over a certain period of time. On the market, the profit Y earned in the future after T years has a lower value X in the present due to the possibility to earn $Y-X$ money through investment in the financial markets with similar risk based on a certain discount rate R . Herein, X is the net present value (NPV), calculated as $Y/(1+R)^T$. Again, this can be regarded as a variation in price and thus value. Changes based on percentages inherently distort the value ratios between the services over time; an identical service has a higher price and ‘value’ in 2010 than in 2011, which implies that the benefit to future mankind is less important than to the current population. However, the unit values of services will change over time due to various factors, and not considering discounting implies that the welfare of the current generation is negligible for long-term future predictions, which is questionable. Many pros and cons can be formulated for discounting; see the specific work of Hepburn and Koundouri (2007) on this issue in the field of forest economics. We considered results without price discounting ($R=0\%$) in this manuscript, but NPV results with a constant discount rate of 2 and 4%, as done by Fürstenau et al. (2007), have been obtained. The same primary conclusions are drawn here as those without discounting, as shown and discussed in section I of the supplementary file. In practice, the ‘ecosystem service’ concept could replace more general ecosystem management principles, e.g., the ‘sustainable forest management’ principles, as a framework for management selection, although they should co-exist for now because not all ecosystem services have been (e)valuated well enough (Quine et al., 2013). In addition to its advantages mentioned in section 1.1, valuation through monetization clearly also has its drawbacks compared to MCA methods. This valuation dilemma could, to a certain extent, be resolved by valuating ecosystem services using environmental impact characterization, resulting in an aggregated set of objective tangible values wherever possible, keeping in mind the difference in finality discussed in the previous section. However, then MCA might still be needed to obtain a single outcome. In general, there is no one ideal approach yet though lessons can be learnt out of the existing ones and integration must be researched.

3.4 Influence of space and time boundaries

As previously noted, the temporal and spatial boundaries influence the results of our assessments. Ideally, all impacts and benefits in space and time should be included. However, for practical reasons, boundaries are set, e.g., the global warming potential is assessed over only 100 years, while the effects of emissions might persist longer. A second important point is that, for freshwater loss in Belgium, the human health impact is 0 DALY m⁻³ according to Pfister et al. (2009). Because of the very high human development index (> 0.88) for Belgium (Pfister et al., 2009), the malnutrition vulnerability induced by a loss in agricultural crops is set equal to zero, since Belgium can easily buy water from elsewhere. From a marginal and local perspective, this is acceptable (*ceteris paribus* principle), but, if large quantities are withdrawn, this will always have a direct effect and will result in a loss in agricultural products on the global market (as water from elsewhere is bought); this may thus affect human health, though possibly not locally. Furthermore, water scarcity may become an important issue in Flanders in the near future (Rekenhof aan het Vlaams Parlement, 2014). Evapotranspiration may also lead to an increase in valuable precipitation on downhill agricultural lands (Keys et al., 2012). These boundaries should be considered and possibly broadened through further research. As discussed by Schaubroeck et al. (2013), the time aspect, e.g., the amount of time carbon dioxide is stored, and the regional differentiated aspect of impact/effect need to be better integrated, including in ecosystem service valuation.

Conclusions

The environmental sustainability assessment of ecosystem management needs further improvement regarding the following aspects: quantification/modeling of flows/ecosystem services, the influence of future environmental change on their quantities and valuation of the damage and benefit of these flows/services. In this study, we improved several aspects in a general manner and exemplified these through application to a Scots pine forest ecosystem. In general, the following aspects were improved: freshwater loss through evapotranspiration is considered as a disservice (note that this consideration is site-dependent), the inflow is compared to the outflow to better quantify pollutant removal/generation services and interannual climate variability is accounted for in environmental change scenarios. We

modeled the Scots pine flows/ecosystem services under different environmental change and management scenarios. Because of the highly detailed process-based modeling effort, the following ecosystem services could be quantified in a relatively high qualitative manner: particulate matter (PM) removal, freshwater loss, CO₂ sequestration, wood production, NO_x emissions, NH₃ uptake and nitrogen pollution/removal. To the best of our knowledge, the considered environmental change scenarios are the most elaborated to date due to the consideration of (site-specific) changes in temperature, precipitation, nitrogen deposition, wind speed, PM concentration and CO₂ concentration. Two environmental sustainability assessment approaches were applied: environmental impact assessment and monetary ecosystem service valuation. These approaches result in outcomes that are more tangible and less subjective than those of multi-criteria decision analysis (with stakeholder opinion). Note however that monetization also has its drawbacks (see section 3.3). Though the monetary ecosystem service valuation is quite commonly applied, we used values specifically obtained for the region, but questioned the use of economic value to characterize ecosystem services. Hence, environmental impact assessment, in combination with MCA if no single outcome is obtained, might be reasonable good alternative and also characterizes the final damage and benefit to other ecosystems, in addition to mankind, on which the effect is only considered for ecosystem service valuation. However, addressing consensus on the integration between these approaches is the next step, as the endpoint (the well-being of both humans and nature or only of humans) of any environmental sustainability assessment approach is still debatable.

In this study, the monetary valuation highlights the likely importance of services provided by the forest, with a total estimated yearly average of 361-1242 euro ha⁻¹ yr⁻¹. PM_{2.5} (< 2.5 µm) removal is the key service, with a projected value of 622-1172 euro ha⁻¹ yr⁻¹. These total values do not represent the total impact or the value (of all services) of the forest because not all aspects are considered. Concerning the environmental impact assessment, with net CO₂ uptake being the most relevant contributing flow, a loss prevention of 0.014-0.029 healthy life years ha⁻¹ yr⁻¹ is calculated.

Both assessment methods favor the use of the least intensive management scenario because it results in higher CO₂ sequestration and PM removal induced by a higher leaf area index. Although the current

focus in the literature is often on assessing as many ecosystem services as possible for large areas, the present case study clearly demonstrates the relevance and important new findings of the thorough study of a specific ecosystem. There are, however, still many prominent limitations to various aspects of this study. It is, after all, a prospective modeling effort, and its outcome should be regarded as an estimation. Further research to improve the assessment is imperative.

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